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Environmental Quality Standards to Protect Identified Uses of Controlled Waters

- Salmonid fisheries
- Cyprinid fisheries
- Migratory fisheries
- Commercial harvesting of marine fish for public consumption
- Commercial harvesting of shellfish for public consumption

Sniffer

SCOTLAND & NORTHERN IRELAND FORUM
FOR ENVIRONMENTAL RESEARCH

WRc plc

NRA Project Record 010/9/N



NRA

National Rivers Authority

**ENVIRONMENTAL QUALITY STANDARDS TO PROTECT IDENTIFIED USES OF
CONTROLLED WATERS**

- Salmonid fisheries
- Cyprinid fisheries
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- Commercial harvesting of marine fish for public consumption
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S Lewis

**Research Contractor:
WRc plc
Henley Road Medmenham
PO Box 16 Marlow
Buckinghamshire SL7 2HD**

**National Rivers Authority
Rivers House Waterside Drive
Almondsbury Bristol BS12 4UD**

Project Record 010/9/N

National Rivers Authority
Rivers House Waterside Drive
Almondsbury Bristol
Avon BS12 2UD

Tel: 0454 624400
Fax: 0454 624409

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This document was produced under R&D Project 010 by:
WRc plc
Henley Road Medmenham
PO Box 16 Marlow
Buckinghamshire SL7 2HD

Tel: 0491 571531
Fax: 0491 579094

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NRA Project Leader

The NRA's Project Leader for R&D Project 010:

E Douglas - Northumbria Region

Additional copies

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EXECUTIVE SUMMARY

This report considers the development of environmental quality standards (EQSs) for the salmonid fishery, cyprinid fishery, migratory fishery, commercial harvesting of marine fish for public consumption and commercial harvesting of shellfish for public consumption uses of controlled surface waters. Previous reports (Agg 1989, Smith *et al* 1991, and Smith and Gulson 1991) have been used to identify those parameters necessary for the maintenance of these five uses. Each water use is considered in a separate section within which identified parameters are discussed and standards proposed, a summary of the proposed standards is presented at the beginning of the relevant section.

For salmonid, cyprinid and migratory fisheries, EQSs for substances in water have been proposed for the protection of these fisheries. For the commercial harvesting of marine fish and shellfish for public consumption uses 'Warning Levels' of substances in waters have been proposed. These 'Warning Levels' have been proposed by considering data on bioaccumulation and food standards and aim to prevent acceptable intake values and concentrations in fish/shellfish flesh exceeding statutory or recommended levels. For the commercial harvesting of marine fish for public consumption it has been concluded that the current EQSs for most List II substances for the protection of salt water life should be adequately stringent to protect this use, however for the commercial harvesting of shellfish for public consumption, these List II EQSs do not appear adequate to protect this use and more stringent 'Warning Levels' have been proposed.

For all five uses considered in this report there has been found to be limited information on a number of the parameters considered and in general for indigenous species, this has been found to be especially so when considering migratory fisheries and the commercial harvesting of marine fish and shellfish.

KEY WORDS

Environmental Quality Objectives, Environmental Quality Standards, Fisheries, Salmonid, Cyprinid, Migratory, Commercial Harvesting, Marine Fish, Shellfish

1. INTRODUCTION

The Water Quality Survey Group (WQSG) of the NRA have proposed 14 use-related environmental quality objectives (EQOs) to apply to controlled waters (Water Quality Survey Group 1990). WRC have been contracted to provide technical support to the WQSG under the NRA R&D Project 010 to recommend appropriate environmental quality standards (EQSs) for each of the proposed use-related objectives. For each use it is necessary to identify the EQSs needed to achieve the related EQO.

Agg (1989) summarised the available standards for the protection of different uses of fresh waters, his work has recently been extended in the form of the 'umbrella' report (Smith *et al* 1991) and a summary produced after consultation with the WQSG (Smith and Gulson 1991). These cover all 14 uses and all five categories of controlled waters, they formulate the most critical standards for the protection of each water use and propose interim standards for parameters for which standards are inadequate or not available. Detailed reports for each water use have or are being undertaken. Smith and Hedgecott (1990) and Gulson (1992) reviewed and critically assessed the standards necessary to protect Basic Amenity, and General and Special Ecosystems.

This report reviews and critically assesses existing standards and parameters, some of which were reviewed in Smith *et al* (1991) and Smith and Gulson (1991), and proposes further standards necessary to protect the 'fisheries' uses: Salmonid Fisheries, Cyprinid Fisheries, Migratory Fisheries, Commercial Harvesting of marine fish for public consumption and Commercial Harvesting of shellfish for public consumption.

For some of these five water uses water quality standards have been laid down in EC Directives, for waters designated under the terms of these directives these standards must be met. However, for waters that are used for one of the above uses, but not EC-designated, alternative values may be given where the EC standards are not considered justifiable. EQSs have also been adopted for 17 List I substances in Daughter directives of the EC Dangerous Substances Directive. Standards are laid down for inland waters, estuaries and coastal waters but these are not use-related and therefore apply to all water uses. For List II substances use-related EQSs have been set nationally and these standards are given in the appropriate sections.

2. SALMONID FISHERIES

2.1 Environmental quality objective

The salmonid fishery environmental quality objective (EQO), given below, is applicable to fresh surface waters only. The definition of the EQO for this use in rivers and lakes is similar, but in the definition, those parts applicable only to rivers are in brackets.

EQO	Explanatory notes
Maintain water quality so as to sustain a breeding population of salmonids appropriate to the physical characteristics (and flow régime) of the water body (where this is not precluded by natural physical barriers)	Appropriate water quality standards and/or magnitude of population is specified as a 'standard'

The EQO for salmonid fisheries is concerned with the part of the life cycle of salmonids when they form resident populations in fresh waters. Hence all aspects of the life cycle, eggs to adults need to be considered. Those parameters necessary for maintaining the above objective have been identified from Agg (1989), the EC Fresh Water Fish Directive (CEC 1978), Smith *et al* (1991) and Smith and Gulson (1991). Standards set or proposed for the UK and other countries are reviewed, together with available relevant literature, the List II substances reviewed have been limited to the following:

ammonia, arsenic, boron, chromium, copper, inorganic lead, inorganic tin, iron, mothproofing substances, nickel, organotins, pH, sulphide, vanadium and zinc.

The data for all the considered parameters are critically assessed and where possible, criteria necessary for maintaining the objective are identified or proposed. Proposed standards, and those applicable to EC-designated salmonid fisheries are summarised in Tables 2.1 and 2.2.

2.2 Chemical parameters

2.2.1 Dissolved oxygen

A number of major reviews which propose criteria for dissolved oxygen (DO) for the protection of salmonid fisheries have been carried out in the last ten years, these include EIFAC (Alabaster and Lloyd 1982), US EPA criteria (1986), Anglian Water Authority (1986), and Stiff *et al* (1990) who recently extensively reviewed the available toxicity and field data to propose EQSs for the protection of non-EC-designated fisheries. All these criteria are outlined and discussed below. The CEC (1978), in the Fresh Water Fish Directive, give mandatory and guideline values for dissolved oxygen, these apply to EC-designated salmonid fisheries and are outlined below.

Table 2.1 Environmental quality standards, proposed and set for the protection of salmonid fisheries

Parameter	Units	Designated fisheries (A)				Non-designated	
		Guideline value	Statistic	Mandatory value	Statistic	value	Statistic
Dissolved oxygen	mg O ₂ /l	9	50P	9	50P	9	50P
		7	100P	6	INV	5	95P
						6	MIN (1)
Nitrite	µg N/l	3	95P	-	-	270	95P (2)
BOD	mg O ₂ /l	3	95P	-	-	4	95P (3)
Phosphorous (total)	mg PO ₄ /l	0.2 (4)	-	-	-	-	-
Chlorine	µg HOCl/l	-	-	5	95P	4	MAX
Ammonia (un-ionized)	µg N/l	4	95P	21	95P	21	95P
(total)		30	95P	780	95P	780	95P (G)
pH (5)	-	-	-	6-9 (6)	95P	6-9	95P
Temperature (difference)	°C	-	-	<1.5	98P	<1.5	MAX
(actual)		-	-	<21.5	98P	20	MAX
(breeding of cold water sp.)		-	-	<10	98P		
Suspended solids (5)	mg/l	25	AA	-	-	25	AA
Arsenic (dissolved)	µg/l	-	-	-	-	50	AA (7)
Boron (total)	mg/l	-	-	-	-	2	AA (7)
Iron (dissolved)	mg/l	-	-	-	-	1	AA (7)
Mothproofing Substances (PCSDs)	µg/l	-	-	-	-	0.05	95P (7)
(sulcofuron)		-	-	-	-	25	95P (7)
(flucofuron)		-	-	-	-	1.	95P (7)
(permethrin)		-	-	-	-	0.01	95P (7)
(cyfluthrin)		-	-	-	-	0.001	95P (7)

.../continued

Table 2.1 (continued)

Parameter	Units	Designated fisheries (A)				Non-designated	
		Guideline value	Statistic	Mandatory value	Statistic	value	Statistic
Organotins TBT	µg/l	-	-	-	-	0.02	MAX (7)
TPT		-	-	-	-	0.02	MAX (7)
Sulphide (as H ₂ S undissociated)	µg/l						
<15°C <5 mg O ₂ /l		-	-	-	-	0.5	AA (8)
<15°C <5 mg O ₂ /l		-	-	-	-	5.0	24A (8)
<15°C >5 mg O ₂ /l		-	-	-	-	1.0	AA (8)
<15°C >5 mg O ₂ /l		-	-	-	-	10.0	24A (8)
>15°C <5 mg O ₂ /l		-	-	-	-	0.25	AA (8)
>15°C <5 mg O ₂ /l		-	-	-	-	2.5	24A (8)
>15°C >5 mg O ₂ /l		-	-	-	-	0.5	AA (8)
>15°C >5 mg O ₂ /l		-	-	-	-	5.0	24A (8)
Inorganic tin (total)	µg/l	-	-	-	-	25	AA (8)
Chromium	} Standards are related to water hardness, see Table 2.2						
Copper							
Lead							
Nickel							
Vanadium							
Zinc							

...../continued

Table 2.1 (continued)

Notes	(1) Minimum concentration in interstitial water for spawning ground in spring and autumn.
	(2) Where chloride concentration varies largely from 10 mg/l standard should be varied according to EIFAC criteria.
	(3) Measured as BOD ₅ (ATU) i.e. oxygen demand without nitrification.
	(4) For lakes; value indicative of the need to reduce eutrophication. In the case of lakes of average depth between 18 and 300 m the following formula could be applied: $L < 10 \cdot z/Tw (1 + Tw)$ where L = loading expressed as mg P per square metre lake surface in one year; Z = mean depth of the lake in meters; Tw = theoretical renewal time of the lake in meters;
	(5) Derogation is allowed in the event of exceptional meteorological or geographical conditions.
	(6) Artificial pH variations with respect to the unaffected values shall not exceed ± 0.5 of a pH unit within the limits falling between 6.0 and 9.0 provided that these variations do not increase the harmfulness of other substances present in the water.
	(7) Standards for these List II substances have been adopted as National EQS values (DoE 1989) and these values must be achieved in all waters receiving discharges.
	(8) Standards for these List II substances have been proposed to DoE but not yet adopted as National EQS values. They are suggested as guidelines for the protection of salmonids to be applied where relevant locally.
	G Guideline.
	I Mandatory.
	P Percentile.
AA	Annual average.
24A	Maximum 24-hour concentration.
INV	Investigation level - when the oxygen concentration falls below this level the cause is to be investigated and the appropriate measures are to be taken.
MAX	Maximum allowable concentration.
MIN	Minimum allowable concentration.

References: A CEC (1978)

Table 2.2 Hardness related EQSs proposed for the protection of salmonid fisheries

Substance	Hardness (mg/l as CaCO ₃)						Notes
	<50	50-100	100-150	150-200	200-250	>250	
Chromium	5	10	20	20	50	50	D, AA (1, 2)
Copper	1	6	10	10	10	28	D, AA (1)
Lead	4	10	10	20	20	20	D, AA (1)
Nickel	50	100	150	150	200	200	D, AA (1, 3)
Vanadium	20	20	20	20	60	60	T, AA (1)
Zinc	8	50	75	75	75	125	T, AA (1)

Notes (1) National EQS value (DoE 1989).
 (2) Concentrations reaching the EQS may caused detrimental effects to early life stages.
 (3) Concentrations reaching EQS of intermediate hardness may cause detrimental effect.
 AA Annual average.
 D Dissolved.
 T Total.

Standards applicable to the UK

The CEC (1978) have set guideline and mandatory dissolved concentrations for areas designated as salmonid fisheries. The mandatory value of ≥ 9 mg O₂/l is for 50% of samples and has the proviso that if the concentration falls below 6 mg O₂/l the cause must be investigated and the appropriate measures adopted. The directive also sets guideline values of ≥ 9 mg O₂/l (50% of samples) and ≥ 7 mg O₂/l (100% of samples).

No national standards for dissolved oxygen have been adopted in the UK for non-EC-designated salmonids fisheries. However, a number of standards have been proposed for the protection of salmonid species in the UK.

EIFAC (Alabaster and Lloyd 1982) have recommended annual 50 and 95 percentile DO values for resident populations of salmonids of greater than 9 mg O₂/l and 5 mg O₂/l respectively. However, they are intended only as guidelines due to seasonal and geographical variations. It states that since early life stages are particularly sensitive the lower DO value should not occur when these stages are present. In addition, EIFAC (Alabaster and Lloyd 1982) have proposed tentative minimum steady state DO criteria for normal successful fulfilment of the fish life-cycle under otherwise favourable conditions, these are shown in Table 2.3.

Table 2.3 Tentative minimum sustained DO for maintaining the normal attributes of the life-cycle of fish under otherwise favourable conditions (Alabaster and Lloyd 1982)

Attribute	DO (mg/l)
Survival of juveniles and adults for one day or longer	3
Fecundity, hatch of eggs, larval survival	5
10% reduction in hatched larval weight	7
Larval growth	5
Juvenile growth (could be reduced by 20%)	4
Growth of juvenile carp (<i>Cyprinus carpio</i>)	3
Cruising swimming speed (maximum sustainable speed could be reduced by 10%)	5
Upstream migration of Pacific salmon (<i>Oncorhynchus spp.</i>) and Atlantic salmon (<i>Salmo salar</i>)	5
Upstream migration of American shad (<i>Alosa sapidissima</i>)	2
Schooling behaviour of American shad	5
Sheltering behaviour of walleye (<i>Stizostedion vitreum vitreum</i>)	6

Anglian Water Authority (1986) adopted criteria for the protection of fisheries in its area. For salmonid fisheries not more than 5% of values (5 percentile) should be below 7 mg/l and 1% of values (1 percentile) below 6 mg/l.

Stiff *et al* (1990) have carried out the most recent review of data on dissolved oxygen. They proposed an EQS, stated to be adequate for the protection of salmonid fisheries, as two standards, a median of dissolved oxygen concentration 9 mg/l and a 95 percentile of ≥ 5 mg/l. However, they mentioned that early life-stages may require more stringent conditions.

Milne and Seager (1990) have proposed tentative criteria for dissolved oxygen which take into account the magnitude, duration and return period of pollution events. While such criteria would be valuable for the prediction of the effects of intermittent pollution events, as mentioned in Smith *et al* (1991) there is still a need for validation in the field and thus they cannot yet be recommended for use when setting EQSs for fisheries.

Standards from other countries

Standards for dissolved oxygen have been proposed by a number of other countries, these are outlined below.

The US EPA (1986) have set criteria for ambient oxygen concentrations for the protection of fresh water aquatic life (Table 2.4). The criteria are derived from production impairment estimates based primarily on growth data and information on temperature, disease and pollutant stresses. Each criterion should be viewed as an estimate of the threshold concentration below which detrimental effects are expected. The criteria represent worst case conditions and not assured no-effect levels, thus levels would be expected to be better than the criteria for most of the time at most sites. The criteria for early life stages are intended to apply only when and where these stages occur and the one day minimum values are considered to be instantaneous concentrations that should be achieved at all times. Criteria for cold water fish apply to waters containing a population of one or more species of the family Salmonidae or fish deemed to be as sensitive as salmonids. Despite the fact that they found the acute lethal limit for salmonids to be 3 mg/l or less, the cold water minimum was set at 4 mg/l due to insects common to salmonid habitats being found to be less tolerant of acute exposures than salmonids. In addition, it is stated that in situations where criteria conditions are just maintained for considerable periods the proposed criteria may represent some risk of production impairment. If such slight production impairment is deemed unacceptable then the 'no production impairment' values (Table 2.5) should be used and those in Table 2.4 should be used as minima.

Table 2.4 US water quality criteria for dissolved oxygen (mg/l) (US EPA 1986)

	Life stage	Cold water Criteria e.g. salmonids	Warm water Criteria e.g. cyprinids
30 day mean	all	6.5	5.5
7 day mean	early (1)	9.5 (2)	6.0
7 day mean minimum	all	5.0	4.0
1 day minimum (3)	all	4.0	3.0
1 day minimum (3)	early (1)	8.0 (2)	5.0

- Notes: (1) Includes all embryonic and larval stages and all juvenile forms to 30 days after hatching.
 (2) These are the water column concentrations recommended to achieve the required intragravel dissolved oxygen concentrations. For species which have early life-stages exposed directly to the water column values 3 mg/l less should be applied.
 (3) All minima should be considered as instantaneous concentrations to be achieved at all times.

Table 2.5 US water quality criteria to be used if slight impairment of production is unacceptable (mg/l) (US EPA 1986)

	Life stage	Cold water Criteria e.g. salmonids	Warm water Criteria e.g. cyprinids
Mean (4)	early	11.0 (2)	6.5
Minimum (5)	early	9.0 (2)	5.5
Mean (4)	all	8.0	6.0
Minimum (5)	all	6.0	5.0

- Notes: (4) Mean values are based on 'no production impairment' values.
 (5) Minima are based on 'slight production impairment' values.

Criteria have been proposed for dissolved oxygen by the National Research Centre for Canada (Davis 1975). These differentiate between fish populations, temperature and levels of risk and are presented in Table 2.6. Three levels of risk are used, A, B and C.

Table 2.6 NRCC dissolved oxygen criteria (Davis 1975)

Group	Protection level (1)	Concentration (mg/l)
Mixed fresh water fish population, including salmonids	A	7.25
	B	5.25
	C	3.25
Mixed fresh water fish population with no salmonids	A	5.50
	B	4.00
	C	2.50
Fresh water salmonid population	A	7.75
	B	6.00
	C	4.25
Salmonid larvae and mature eggs of salmonids	A	9.75
	B	8.00
	C	6.50
Marine non-anadromous species	A	8.75
	B	6.75
	C	4.50
Marine anadromous species including salmonids	A	9.00
	B	6.50
	C	4.00
Anadromous salmonids in sea water	A	9.25
	B	7.00
	C	4.50

Note: (1) See text for discussion of protection

The criteria are based on mean incipient thresholds for documented sublethal effects:

Level A is the level at which few members of the fish community will be likely to exhibit low DO effects (it represents more or less ideal conditions; mean incipient threshold plus one standard deviation).

Level B is the level at which average members of the community may begin to exhibit symptoms of oxygen distress (mean incipient threshold). Some degree of risk to a portion of the population exists at this level if the oxygen minimum period is prolonged beyond a few hours.

Level C is the level at which a large proportion of the population may be affected by low DO. It should only be applied to hardy fish populations or fisheries of marginal economic importance and is based on the mean incipient threshold minus one standard deviation.

Discussion

The standards outlined above are based on reviews of the effects of low DO concentrations on salmonids, the most recent of which was Stiff *et al* (1990). Toxicity data have shown adult trout to survive at DO levels between 2 and 3 mg/l for three to four days (Alabaster *et al* 1979), but that swimming speeds of salmon and trout were impaired at concentrations less than 5 mg/l (Chapman 1986 citing a number of studies). Chapman (1986) has also cited laboratory studies which showed adult chinook and coho salmon avoided DO concentrations of 3 mg/l and less, sometimes even concentrations as high as 6 mg/l. The particular susceptibility of early life stages has been shown by Siefert *et al* (1974). Severe mortalities of embryos occurring below 3 mg/l, the no-effect level for embryonic and larval survival and development seems to be approximately 6 mg/l. Stiff *et al* (1990) mentioned that early life-stages may require more stringent conditions due to eggs being laid in redds made in the gravel where the interstitial DO may be up to 3 mg/l lower than in the main body of the water. They expanded the minimum criteria of DO for maintaining the normal attributes of the life-cycle of fish (Table 2.3) by EIFAC (Alabaster and Lloyd 1982) to include other attributes of the life cycle of salmonid fish, including dissolved oxygen requirements for the interstitial water of gravel beds (Table 2.7).

Table 2.7 Minimum sustained DO for maintaining the normal attributes of the life-cycle of fish without significant impairment (Stiff *et al* 1990)

Attribute	DO (mg/l)	
	Salmonid	Non-salmonid
Spawning	-	5
Hatching, embryo survival	7*	6
Hatched larval weight	-	7
Larval growth	6*	5
Larval survival for a few hours	-	3
Larval survival, longer term	6*	5
Juvenile growth	6	5
Juvenile survival for a few hours	3	3
Juvenile survival, longer term	5	-
Adult survival, for a few hours	2	-
Adult survival, longer term	4	3
Physiological stress	-	5
Physiological stress, adult	5	4
Swimming speed, juvenile	6	-
Swimming speed, adult	5	3
Behaviour, adult	-	6
Estuarine migration, salmon smolts	5	-

Note: * Refers to DO requirements for the interstitial water of the gravel beds.

Field data cited in Stiff *et al* (1990) from UK National River Authority regions and River Purification Boards on dissolved oxygen concentrations in UK rivers showed salmonid fisheries to be recorded at 95 percentile concentrations of 7 to 8 mg/l dissolved oxygen.

For the maintenance of a resident population of salmonids, any criteria must be sufficient to maintain all life stages, from eggs to adults. In addition, it must be considered that early life stages will be situated in 'redds' where interstitial oxygen concentrations are lower than the surrounding water. Stiff *et al* (1990) and US EPA (1986) suggested interstitial oxygen concentration in redds may be 3 mg/l lower than in the surrounding water column. Thus, when considering a dissolved oxygen standard for resident populations of salmonids, the standards must be adequate to protect the stages at these lower oxygen concentrations. The criteria and data reviewed above have given lower dissolved oxygen concentration levels at which early life stages can survive.

The US EPA (1986) stated 6 mg/l was the limit to avoid acute mortality, while Davis (1975) found the status of salmonid larvae and mature eggs would be poor at dissolved oxygen concentrations of 6.5 mg/l. Dissolved oxygen concentrations at which early life stages have been found to be adequately protected have also been reported. Stiff *et al* (1990) reported larval survival and growth at 6 mg/l (interstitial oxygen concentration), Siefert *et al* (1974) found no effect on early life stages and larvae at 6 mg/l and Davis (1975) reported a good salmonid fishery at 7.75 mg/l and a good status of salmonid larvae and mature eggs at 9.75 mg/l.

Thus it is proposed that standards that would adequately protect salmonids fisheries would be a 50 percentile of 9 mg/l and a 95 percentile of 5 mg/l. These standards, which have been previously proposed, fit the available data well and appear to adequately protect salmonid populations. However, in addition a guideline minimum DO concentration of 6 mg/l in interstitial water is suggested to protect spawning grounds. This should apply from autumn to spring and is based on the 'no-effect' level reported by Siefert *et al* (1974).

2.2.2 Nitrite

Nitrite (NO_2^-) (an intermediate oxidation state between ammonium and nitrate) is a naturally occurring anion both in fresh and marine waters. Its typical concentration in oxygenated waters being less than 0.005 mg/l NO_2 (Lewis and Morris 1986).

Eddy *et al* (1983) cited in Lewis and Morris (1986) stated that nitrite ions are actively taken up by most fresh water fish, and that concentrations in the blood plasma may reach ten times that of the surrounding medium. From the blood plasma, nitrite diffuses into red blood cells where it oxidises the iron in haemoglobin to the 3+ oxidation state, forming methaemoglobin, which lacks the capacity to bind oxygen reversibly. Reductase converts methaemoglobin back to haemoglobin, but this is slow, the normal proportion of haemoglobin being restored in 24 - 48 hours on transferral to water lacking nitrite (Huey *et al* 1980, cited in Lewis and Morris 1986).

Standards applicable to the UK

The EC (CEC 1978) has set a guideline standard for nitrite in EC-designated salmonid fisheries of $<0.01 \text{ mg NO}_2/\text{l}$ as a 95 percentile, this is equivalent to $0.003 \text{ mg NO}_2\text{-N}/\text{l}$.

No environmental quality standards for the protection of non-EC-designated fisheries have been adopted in the UK. However, EIFAC (1984) reviewed toxicity data and proposed tentative criteria (95 percentile and annual average concentrations) for the protection of salmonid fish (Table 2.8). In addition, they noted that nitrite is most toxic to fresh water fish at low concentration of chloride, and thus the criteria were proposed with respect to the concentration of chloride present.

Table 2.8 Tentative nitrite criteria for the protection of salmonid ($\text{mg/l NO}_2\text{-N}$; EIFAC 1984)

Chloride (mg/l)	Nitrite ($\text{mg/l NO}_2\text{-N}$)	
	average	95 percentile
1	0.01	0.03
5	0.05	0.15
10	0.09	0.27
20	0.12	0.36
40	0.15	0.45

Standards from other countries

Both the US EPA (1986) and CCREM (1987) have recommended that for the protection of salmonid fish levels of nitrite should be at or below $0.06 \text{ mg/l NO}_2\text{-N}$.

Discussion

New toxicity data on the effects of nitrite on salmonids, since EIFAC (1984) is limited, the majority of studies reported in EIFAC (1984) were on salmonids, particularly rainbow trout, with little work being done on coarse fisheries.

Both the US EPA (1986) and CCREM (1987) cited a study by Russo *et al* (1974) on which they base their standard for the protection of fresh water aquatic life. Russo *et al* (1974) found no mortality of rainbow trout occurred over 10 days exposure to nitrite at 0.06 mg/l . However, the CCREM also cited Wedemeyer and Yasutake (1978) who found that steelhead juveniles exposed for six months to 0.06 mg/l showed no reduction in growth at a chloride concentration of 2.3 mg/l but gill damage did occur.

EIFAC (1984) noted that nitrite toxicity is strongly alleviated by chloride ions. They considered that the concentration of chloride relative to that of nitrite (expressed by weight

mg Cl⁻ mg/l: NO₂-N mg/l) was important when considering water quality and toxicity to fish. For rainbow trout they concluded that maximum protection would be at a weight ratio of 17. Bearing in mind the important effect of chloride, EIFAC (1984) concluded that salmonids are less resistant to nitrite than coarse fish and other related species, while fry and fingerlings of salmonid species are more tolerant than larger fish. In addition, Lewis and Morris (1986) concluded that small fish including larval stages are unlikely to be more sensitive to nitrite than larger fish of the same species.

EIFAC (1984) also concluded that the toxicity data with respect to fresh water fish can only be correctly assessed with other water quality values, such as chloride, calcium and pH. pH appeared to be an important variable, but lack of data especially at intermediate pH values did not allow any firm conclusions to be made.

A survey by Solbe (1981) reported by EIFAC (1984) showed that in waters with mean chloride concentrations up to 25 mg/l, good salmonid fisheries were associated with nitrite concentrations below 50 µg NO₂-N/l. In addition, the US EPA (1986) stated that the concentrations of nitrite that would exhibit toxic effects on cold water fish (salmonids) were unlikely to occur in natural surface water in the US and thus did not recommend restrictive criteria. Annual mean chloride concentrations in UK rivers have been reported to be in the range 3 to 100 mg/l (Gardiner and Smith 1990), with typical values being 10 to 30 mg/l except in the Severn Trent and Thames regions where the values were higher.

The standard of 0.06 mg/l proposed by the US EPA (1986) and the CCREM (1987) while low, has been shown to cause sub-lethal effects in salmonids. However, details of chloride concentrations, which are known to reduce toxicity, are not given. In addition, the standards set by the EC (CEC 1978) for the protection of EC-designated salmonid fisheries are ten times lower than proposed by EIFAC, and the scientific justification for this standard cannot be ascertained.

Field data cited in EIFAC (1984) indicate that salmonid fisheries will be adequately protected at the levels proposed in their report. Hence it is proposed (as previously done so by Smith *et al* 1991), in view of chloride data from Gardiner and Smith (1990), that the EIFAC criteria for nitrite at 10 mg Cl/l, i.e. 0.27 mg NO₂-N/l as a 95 percentile will adequately protect salmonid fisheries. Where chloride concentrations deviate widely from 10 mg Cl/l, the standard should vary according to the EIFAC guidelines.

2.2.3 Biochemical oxygen demand

Biochemical oxygen demand (BOD) is an indicator of the oxygen demand of the organic loading in a watercourse rather than being of direct toxicity to fish. This is often measured after a period of five days, hence BOD₅. If the measurement is conducted in the presence of allyl thiourea (ATU; a nitrification inhibitor) then the oxygen demand is measured without nitrification.

Standards applicable to the UK

The EC Fresh Water Fish Directive (CEC 1978) has a guideline standard of 3 mg O₂/l as a 95 percentile for the protection of designated salmonids fisheries. No standards have been

adopted in the UK for non-EC-designated salmonid fisheries, although a number of bodies have proposed standards that are relevant, but all are less stringent than the EC criteria.

Anglian Water Authority (AWA) (1986) proposed 95 and 99 percentile standards of 4.0 and 6.0 mg O₂/l respectively for salmonid dominated rivers. These criteria are expressed as BOD₅ (ATU) which is oxygen demand without nitrification and thus is slightly less stringent than BOD₅. In addition, AWA state that higher 95 percentile BOD₅ (ATU) of up to 6.0 mg O₂/l are known to occur in some good quality salmonid dominated fisheries, and so indicate scope for local criteria in some cases.

A standard for BOD₅ of 5 mg O₂/l has been proposed by Baudo *et al* (1978) and 5 mg O₂/l and 3 mg O₂/l as a mandatory and guideline value were proposed by Chiaudani and Premazzi (1988). However, both Baudo *et al* and Chiaudani and Premazzi were criticised in Smith *et al* (1991) for not giving the derivation or statistical requirements of the standards.

Discussion

All these standards have been previously mentioned in Smith *et al* (1991) and additional information since then is not available. Smith *et al* (1991) proposed a guideline standards of 95 percentile 4 mg O₂/l based on the standards set by former Anglian Water Authority (AWA 1986) with a proviso allowing higher values where there is little risk of reducing DO levels below the relevant criteria. Since no further information is available it is proposed that this standard, 95 percentile 4 mg O₂/l expressed as BOD₅ (ATU) be used as a guideline for the protection of salmonid fisheries, along with a proviso allowing higher values where these are not found to be detrimental to the fishery.

2.2.4 Phosphorus

A standard for phosphate concentrations for shallow lakes which 'may be regarded as indicative in order to reduce eutrophication', has been set for EC-designated fisheries in the Fisheries Directive (CEC 1978). In addition, a formula for the calculation of maximum acceptable loading for deeper lakes is given. However, the standards are set in order to reduce eutrophication and not for the protection of salmonids against toxic effects of phosphate. Phosphate is not directly toxic to fish, rather effects of phosphate to fish are through reduced oxygen concentration that may occur due to algal blooms. It is therefore concluded that since eutrophication is likely to have detrimental effects on other water uses before fisheries, that a standard for the protection of salmonid fisheries is not necessary.

2.2.5 Chlorine

In water chlorine undergoes a pH-dependant reaction to produce hypochlorous acid (HOCl) which, together with Cl₂, comprises 'free chlorine'. Additional reactions with amines and ammonia result in the formation of chloramines or 'combined chlorine'. Thus the 'total (residual, available) chlorine' is comprised of 'free' and 'combined' chlorine. EIFAC (Alabaster and Lloyd 1982) considered that the toxic chemical species in solution in chlorine poisoning is hypochlorous acid and thus the toxicity of any solution containing chlorine will depend on the pH of the solution as this will markedly influence the degree of dissociation.

Standards applicable to the UK

Standards for chlorine for the protection of fresh water fish have been set by the EC, (CEC 1978) and apply for designated salmonid fisheries.

The CEC (1978) Fresh Water Fish Directive gives a mandatory 95 percentile concentration of $<5 \mu\text{g/l}$ total residual chlorine (as HOCl) at pH 6 for EC-designated salmonid fisheries, higher concentrations of total chlorine can be accepted if the pH is higher. Sampling frequency is specified as monthly. However, it should be noted, that the Directive expresses the standard as 'total residual chlorine', but this is contradicted by stating that it should be expressed as HOCl. It is probable that the standard refers to HOCl.

EIFAC (Alabaster and Lloyd 1982) carried out a review of the toxicity of chlorine to European fish and recommended a maximum concentration of $4 \mu\text{g/l}$ hypochlorous acid (HOCl). This maximum amount of $4 \mu\text{g HOCl/l}$ corresponds to various concentrations of total chlorine according to the pH or temperature. These are given in Table 2.9.

Table 2.9 Approximate total concentration of chlorine corresponding to $4 \mu\text{g HOCl/l}$ (Alabaster and Lloyd 1982)

Temperature (°C)	Concentration ($\mu\text{g/l}$)			
	pH 6	pH 7	pH 8	pH 9
5	4	5	11	75
25	4	5	16	121

Standards from other countries

Both the US EPA (1986) and the CCREM, (1987) have recommended chlorine criteria for the protection of aquatic life.

The CCREM (1987), recommend that for the protection of fresh water aquatic life the total amount of residual chlorine should not exceed a concentration of $2 \mu\text{g/l}$.

The US EPA (1986) concluded that for the protection of fresh water organisms the four day average concentration of total residual chlorine should not exceed $11 \mu\text{g/l}$ more than once every three years on average and the one hour average concentration should not exceed $19 \mu\text{g/l}$ more than once every three years.

Discussion

The EIFAC (Alabaster and Lloyd 1982) guidelines are based on a review of the toxicity of chlorine to European fresh water fish. The authors concluded that concentrations greater than $8 \mu\text{g HOCl/l}$ were harmful or lethal within four days to salmonids or coarse fish. While

concentrations of 4 µg HOCl/l or less could be generally endured by sensitive species they stated that this may result in some mortality. However, since chlorine is reactive and does not persist for long in most streams, they believed the upper limit concentration could be set closer to lethal limits than for other chemicals.

In reviewing the available literature the CCREM (1987) cited Arthur and Eaton (1971) and Arthur *et al* (1975) who reported concentrations of 3.7 and 3.4 µg/l total residual chlorine caused chronic toxicity to *Daphnia sp.* and *Gammarus pseudolimnaeus*. In addition, Arthur *et al* (1975) found 11.22 µg/l total residual chlorine caused chronic toxicity to fathead minnow in life-cycle tests.

The US EPA have recently changed their water quality criteria from 2.0 µg/l total residual chlorine for salmonid fish and 10 µg/l for other fresh water organisms. The new US EPA criteria (US EPA 1986) are based on a review of data on the toxicity of total residual chlorine to 33 fresh water species in 28 genera. From this review they concluded that adequate protection of fresh water organisms would be obtained if the four day average concentration of total residual chlorine did not exceed 11 µg/l more than once every three years on average and the one hour average concentration did not exceed 19 µg/l more than once every three years.

A review of the recent literature shows that there appears to be no additional data on the toxicity of chlorine to fresh water salmonids that has not been discussed in one of the reviews mentioned above.

Standards for the protection of fish have been set as HOCl by EIFAC (Alabaster and Lloyd 1982) and the CEC (1978), while those in other countries, for the protection of aquatic life, they have been set as total chlorine. EIFAC (Alabaster and Lloyd 1982) proposed a limit of 4 µg HOCl/l which was considered to be close to the limit of toxicity to fish. The limit of 5 µg HOCl/l for EC-designated fisheries (CEC 1978) is similar to that proposed by EIFAC, but the scientific conclusions on which it is based are not available. The CCREM (1987) standard of total residual chlorine not to exceed 2 µg/l is stringent, due to the consideration of the toxicity studies by Arthur and Eaton (1971) and Arthur *et al* (1975) as mentioned above. However, the US EPA (1986) criteria are far higher than the CCREM criteria and exceed the values quoted for the toxicity tests in this criteria. In addition, when considering available HOCl at different pHs, the concentration of HOCl at both pH 6 and 7 or higher, for the US EPA (1986) one and four day average concentrations of total chlorine, is likely to exceed or be close to 8 µg/l, a concentration found by EIFAC to be toxic to salmonids and coarse fish.

It is therefore proposed that an EQS of 4 µg HOCl/l, expressed as a Maximum Allowable Concentration (as proposed by EIFAC), will adequately protect non-EC-designated salmonid fisheries. The criteria suggested by the CCREM (1987) and the US EPA (1986) appear too stringent and too lax, respectively, for the protection of UK indigenous fish.

2.2.6 Ammonia

Standards applicable to the UK

Currently the only ammonia standards which apply for fisheries in the UK are those set by the EC (CEC 1978). Standards have been proposed by EIFAC, (Alabaster and Lloyd 1982) and by Seager *et al* (1988), which may apply to UK fisheries, however neither have been adopted nationally.

The Fresh Water Fish Directive (CEC 1978) sets standards for both total ammonia (NH_4^+) and un-ionized ammonia (NH_3) for the protection of designated salmonid fisheries, in both cases guideline and mandatory levels are set, and are expressed as 95 percentiles.

For un-ionized ammonia a guideline value of 0.005 mg NH_3/l (equivalent to 0.004 mg $\text{NH}_3\text{-N}/\text{l}$) and a mandatory level of 0.025 mg NH_3/l (equivalent to 0.021 mg $\text{NH}_3\text{-N}/\text{l}$) are set. A minimum sampling frequency of every month is given, however values may be exceeded in the form of minor peaks in the daytime.

In addition, to un-ionized ammonia standards, the Directive also set standards for total ammonia "in order to diminish the risk of toxicity due to non-ionized ammonia, of oxygen consumption due to nitrification and of eutrophication, the concentrations of total ammonium should not exceed" the standards set. Again both guideline and mandatory values, expressed as 95 percentile with a minimum sampling frequency of once per month are set. The guideline value is 0.04 mg NH_4/l (equivalent to 0.03 mg N/l) and the mandatory 1 mg NH_4/l (equivalent to 0.780 mg N/l). However, the Directive also states that "in particular geographical or climatic conditions and in particular in cases of low water temperature and of reduced nitrification or where the competent authority can prove that there are no harmful consequences for the balanced development of the fish population, member states may fix values higher than 1 mg/l".

EIFAC, (Alabaster and Lloyd 1982) reviewed the data on ammonia toxicity and stated that a healthy salmonid population would be expected to occur where the 95 percentile concentration was below 0.025 mg NH_3/l . They acknowledged that short periods of higher concentrations would be harmless to the fish if they became acclimatised to them and they mention the variation of NH_3 with respect to temperature and pH, presenting a table of values of total ammonia which at a specific pH and temperature will result in an un-ionized ammonia concentration of 0.025 mg NH_3/l .

Seager *et al* (1988) carried out a major review of literature and field data for ammonia. They stated that there was insufficient evidence to justify separate standards for salmonids and cyprinids and proposed an EQS for the protection of all fresh water fish in non-EC-designated water of 0.018 mg NH_3/l (equivalent to 0.015 mg $\text{NH}_3\text{-N}/\text{l}$) expressed as an annual average.

Standards from other countries

The EPA (1986) has set water criteria standards for the protection of fresh water salmonids and other sensitive cold water species. The criteria use formulae which take into account

both pH and temperature to calculate values for one hour and four day exposure to un-ionized and total ammonia. They are presented as a range of acute and chronic limits corresponding to specific combinations of temperature and pH. The concentrations are permitted to exceed numerical limits by an unspecified amount once every three years on average.

Recent data

Since Seager *et al* (1988) only a few studies on the toxicity of ammonia to salmonids have been carried out.

Chronic toxicity of ammonia to salmonids was investigated by Solbe and Shurben (1989), reported as unpublished data in Seager *et al* (1988). They investigated the toxicity of ammonia to early life stages of rainbow trout. Exposure within 24 hours of fertilisation and for the following 73 days caused severe (>70%) mortality, particularly in the eggs at concentrations of NH_3 as low as 0.027 mg/l (0.022 mg $\text{NH}_3\text{-N/l}$). However, when initial exposure was at a later developmental stage, the eye-egg stage (c. 24 day), a decrease in mortality to 40% of the eggs, yolk-sac fry and fry was seen at the same concentration. The authors calculated a tentative LC50 of 0.014 mg $\text{NH}_3\text{-N/l}$ from these results.

Wilson *et al* (1991) found the 14 day LC50 for un-ionized ammonia for the indigenous fish brown trout (*Salmo trutta*) equal to 0.41 mg $\text{NH}_3\text{-N/l}$. No effects on growth were found at concentrations below those found to be lethal, however, the authors believed the length of exposure may have been insufficient for adverse effects to become apparent. In addition, they cited Milne *et al* (1990), who found a similar LC50 for the common but not indigenous rainbow trout, under similar conditions.

Field data (Seager *et al* 1988) show that salmonid fisheries exist at ammonia levels exceeding those proposed by EIFAC (Alabaster and Lloyd 1982). Brown trout were found to have re-colonised a river where mean and maximum ammonia concentrations of 0.026 and 0.077 mg $\text{NH}_3\text{-N/l}$ were reported. They were also reported where mean and maximum un-ionized levels were 0.035 and 0.108 mg $\text{NH}_3\text{-N/l}$, however, the status of the fishery was not known. Data from Anglian Water (Seager *et al* 1988) shows that good and moderate fisheries exist where the 95 percentile un-ionized ammonia concentration exceeds 0.021 mg $\text{NH}_3\text{-N/l}$.

Seager *et al* (1988) reviewed the influence that pH and temperature exert on the toxicity of NH_3 to fish. They concluded that the exact nature of their influence was not sufficiently understood to derive standards that vary according to pH and temperature, as derived by the US EPA (US EPA 1986). Servizi and Gordon (1990) found that the combined toxicity of suspended solids and ammonia was less than expected when their separate toxicities were added. This may indicate ammonia adsorption onto suspended solids.

Discussion

The data has indicated that ammonia may be most toxic to early life stages of fish during chronic exposure. Solbe and Shurben (1989), showed that severe effects on rainbow trout eggs can be demonstrated in waters containing concentrations of un-ionized ammonia at

least as low as 0.027 mg NH₃/l (0.022 mg N/l). This result was found to be much lower than other tests reported in Seager *et al* (1988). The result erodes the safety factor incorporated by EIFAC (at least for rainbow trout) when proposing the tentative standard of 0.025 mg/l as NH₃ for all species of European fish, and led Seager *et al* (1988) to propose an annual average EQS of 0.015 mg NH₃-N/l.

However, few tests on the toxicity of ammonia to early life stages of salmonids have been carried out, and while Solbe and Shurben (1989) indicate early life stages to be more sensitive than adults, other toxicity data shows this not to be necessarily so. A number of factors such as pH, temperature, and suspended solids may influence toxicity as well as the fact that un-ionized ammonia is rapidly and spontaneously converted into a non-toxic form, Lewis (1988).

The US EPA (1986) based their ammonia standards on the influence of pH and temperature on toxicity. Lewis (1988) reviewed the US EPA criteria and the incorporation of pH and temperature dependency. He stated that while some experiments have demonstrated changes in the toxicity of specific concentrations of un-ionized ammonia in relation to ambient pH and temperature other have failed to show such a dependency. While, acknowledging that the inclusion of pH and temperature into the guidelines make the criteria more realistic, he stated that such complex criteria are difficult to evaluate with the probability that new errors may be incorporated, and sources of old errors may become obscure. Lewis (1988) concluded that a more complete treatment of error and an improved database (since the curve fitting was based on four species) was desirable before this method was used. In addition, Mallet (1990) reviewed the criteria and concluded that application of the EPA model to UK waters could not be justified without a better understanding of the influence of pH and temperature on the toxicity of un-ionized ammonia to indigenous species.

With such influences on its toxicity, field data helps to give a much better understanding of the effect of ammonia in the environment. As previously indicated, data reported by Seager *et al* (1988) indicate that good salmonid fisheries occur where the 95 percentile exceeds 0.021 mg NH₃-N/l.

Therefore, based on field data, it is proposed that salmonid fisheries will be adequately protected at a 95 percentile concentration of less than 0.021 mg NH₃-N/l, this is the same as the mandatory level set by the EC (CEC 1978), and that proposed by EIFAC for the protection of salmonid fisheries. However, higher levels should be allowed if found not to have a deleterious effect.

Smith and Gulson (1991) discussed the need for standards for total ammonia to provide a firm basis for the calculation of discharge consents. They concluded that due to differences in ambient pH and temperature characteristics between rivers further research was needed before a meaningful standard which takes into account the pH and temperature of the receiving waters for total ammonia could be set. They suggested that the mandatory standard for EC-designated fisheries could be adopted as an interim standard. It is therefore proposed that this standard, 1 mg NH₄/l (equivalent to 0.780 mg N/l) expressed as a guideline be used for the protection of salmonid fisheries, with a proviso allowing higher values where it can be shown that there are no deleterious effects on salmonid populations. It is suggested that further work be carried out on models predicting the effects of temperature and pH on the toxicity of ammonia, so that more adequate standards can be set.

2.2.7 Arsenic

Arsenic (As) may be released into water from mining, smelting and refining and other processes either indirectly (e.g. leaching from mine workings or slag heaps) or directly. Arsenic is also used as a wood preservative, in the manufacture of glass, alloys or medicines.

Standards applicable to the UK

Mance *et al* (1984a) proposed an environmental quality standard of 50 µg As/l as a dissolved annual average concentration. This has since been adopted by the Department of the Environment (DoE 1989) for the protection of sensitive fresh water life (e.g. salmonids). Despite information that the trivalent form may be more toxic to aquatic life than the pentavalent form, Mance *et al* (1984a) did not suggest separate standards because a meaningful comparison of the two forms was not possible. A review of the EQS is currently being undertaken by WRc for the Department of the Environment.

Standards from other countries

The US EPA (1986) recommend standards for trivalent arsenic for the protection of fresh water aquatic organisms. They state that aquatic life should not be affected unacceptably if four day average concentrations of arsenic (III) do not exceed 190 µg/l more than once every three years on average and if the one hour average concentration does not exceed 360 µg/l more than once every three years.

The CCREM (1987) recommend a guideline of 0.05 mg/l total arsenic for the protection of aquatic life. The standard is stated to be well below concentrations known to be toxic to sensitive early life stages.

Recent data

Mance *et al* (1984a) found a meaningful comparison of the trivalent and pentavalent forms of arsenic was not possible and that there was no evidence for variation of toxicity of arsenic to aquatic organisms in relation to water hardness. The CCREM (1987) also found water hardness did not effect toxicity of arsenic to fish, but high water temperatures appear to increase toxicity.

Data since Mance *et al* (1984a), outlined below, do not indicate a greater sensitivity of fresh water fish than reported previously.

Reviewing the available data on arsenic toxicity, the CCREM (1987) stated concentrations of arsenic (III) causing acute toxic responses in fish ranged upwards of 13.34 mg/l (fingerling rainbow trout, National Fisheries Research Laboratory 1980) and 14.96 mg/l (adult brook trout, Cardwell *et al* 1976). For arsenic (V), 10.8 mg/l for rainbow trout (Hale 1977) was cited as the lowest known acutely toxic concentration.

McGeachy and Dixon (1989), found rainbow trout (*Oncorhynchus mykiss*) to be more sensitive to arsenite (trivalent) than to arsenate (pentavalent), the sensitivity to arsenate was temperature dependent. For arsenite, the 144 hour LC50s were 17.7 at 5°C and 20.7 mg As/l at 15°C, while for arsenate they were 114 and 58.0 mg As/l at 5 and 15°C respectively. The

authors suggested the increase in toxicity may be due to an increase in the rate of arsenate uptake. Increased toxicity was also found by Mallet (1990) for brown trout. The acute toxicity of arsenite (III) was nearly an order of magnitude more toxic to brown trout than then arsenate (V). The 96 hour LC50 for arsenate was in the range 60 - 85 mg/l while for arsenite it was 11.0 mg/l. In addition, the threshold for arsenite had still not been reached after 96 hours (i.e. the LC50 was still decreasing).

Chronic tests, also carried out by Mallet (1990) on brown trout, showed no effects on growth after eight weeks exposure to 0.1 mg As(V)/l.

Reviewing the literature on the toxicity of arsenic to fresh water fish, Mallet (1990) found that the majority of published data is for American species. He cited two tests carried out on the non-indigenous salmonid rainbow trout. Investigating arsenic toxicity to early life stages of rainbow trout over a 60 day period on newly-hatched to swim-up fry, Mallet (1987) found no adverse effects at concentrations up to and including 0.63 mg As(III)/l. While, Mallet (1986), found the 21 day LC50 for As (III) was 5.7 mg/l for fingerling rainbow trout, no effects were seen at 1.0 mg/l, but after 21 days 10% mortality occurred at 3.0 mg/l.

Concentrations of arsenic in UK fresh waters (Mance *et al* 1984a) suggested that the EQS for the protection of fresh water fish would only be likely to be exceeded in a few places none of which currently supports fish life.

Discussion

Toxicity data since the review by Mance *et al* (1984a), including the review of data in CCREM (1987) do not indicate arsenic, in either the trivalent or pentavalent form, to be of greater toxicity than previously found. It therefore appears that the adopted EQS of 50 µg/l (dissolved annual average) arsenic is sufficiently strict to adequately protect salmonid fisheries.

2.2.8 Boron

Standards applicable to the UK

Mance *et al* (1988a) proposed an environmental quality standard of 2 mg/l as an annual average for total boron (B), for the protection of fresh water fish, irrespective of water hardness. This has since been adopted by the Department of the Environment (DoE 1989) for the protection of sensitive aquatic life (salmonids).

Standards from other countries

No other country appears to have set or proposed standards for boron for the protection of fresh water fish or all aquatic organisms.

Discussion

The EQS of 2 mg B/l proposed by Mance *et al* (1988a) was based on limited data (only five studies involving seven species, only two of which were salmonid species) and no reliable data on indigenous fish. Based on these data, Mance *et al* (1988a) found it was not possible to propose a separate standard for salmonid fisheries.

Toxicity data in Mance *et al* (1988a) showed boron toxicity to fish appears to vary according to chemical form. Due to the limited data they were unable to establish any relationship of boron toxicity with respect to water hardness, or different life-stages of fish. However, toxicity appeared to be greater to immature life-stages than to adults.

Additional data since Mance *et al* (1988a) on boron toxicity to salmonid species are very limited. Butterwick *et al* (1989) reviewed the safety of boron in aquatic and terrestrial environments and included a review of the available data on the toxicity of boron to fish. However, the majority of tests cited were used in the derivation of the EQS by Mance *et al* (1988a). Two studies, Procter and Gamble (unpublished) and Birge and Black (1981), not previously cited are given below.

Unpublished data from Procter and Gamble showed the 36 day no observed effect concentration (NOEC) and lowest observed effect concentration (LOEC) for Rainbow trout exposed in natural water were 0.75 and 1.0 mg B/l. However, in well water (27 mg CaCO₃/l) the 60 day LOEC was >17 mg B/l. In addition, Birge and Black (1981), using freshly fertilised rainbow trout eggs found the 32 day NOEC and LOEC in reconstituted hard water (200 mg CaCO₃) were 0.01 - 0.1 mg B/l.

Hamilton and Buhl (1990) investigated the toxicity of boron to swim-up and advanced fry of chinook salmon (*Oncorhynchus tshawytscha*) and coho salmon (*O. kisutch*). Boron was found to be relatively non-toxic with the 96 hour LC50 for both life stages for both fish being greater than 100 mg/l.

Field data from Bingham (1982) cited in Butterwick *et al* (1989) indicated healthy early life stage rainbow trout in surface waters containing up to 13 mg B/l. In addition, concentrations of boron in UK surface waters (Mance *et al* 1988a) indicated that the proposed EQS was unlikely to be exceeded.

New data since Mance *et al* (1988a) for salmonids have been limited to non-indigenous fish. However, the data indicate wide differences in toxicities. The only new acute toxicity data reported, Hamilton and Buhl (1990), indicate that these species would be adequately protected against acute toxicity by the EQS. However, the reported chronic data indicate that toxic effects of boron on fish may occur as low as 0.1 mg/l, 20 times lower than the EQS. However, the wide range in chronic results, (reported values 0.1 to >17 mg/l for 32 and 60 day tests), indicates that further investigations need to be carried out on the chronic toxicity of boron to fish. Since field data indicate that levels in UK surface water rarely reach 2 mg/l and a healthy rainbow trout population has been reported at 13 mg/l it is concluded that the EQS for the protection of salmonid fisheries is adequate.

2.2.9 Chromium

Standards applicable to the UK

Mance *et al* (1984b) proposed EQSs for the protection of salmonid fisheries for chromium VI as dissolved annual averages in relation to water hardness. These have since been adopted by the DoE (1989) as standards for both chromium III (Cr III) and VI (Cr VI) for the protection of sensitive (salmonid) aquatic life and are given in Table 2.10. EQSs for chromium III were not proposed by Mance *et al* (1984b) due to lack of data.

Table 2.10 EQSs for chromium (III and VI) for the protection of salmonid fisheries (DoE 1989)

Water hardness (mg/l CaCO ₃)	Dissolved Cr (annual average)(µg Cr/l)
Less than 50	5
50 - 100	10
100 - 200	20
Greater than 200	50

The EQSs are based on data for species that are not indigenous to the UK and hence may be more stringent than necessary. The standards are currently being reviewed by WRc for the Department of the Environment.

EIFAC (1983) reviewed the available data on chromium toxicity to European fresh water fish and concluded that for the protection of salmonid fish, the mean concentration of soluble chromium should not exceed 0.025 mg/l, and the 95 percentile should not exceed 0.1 mg/l. They stated that in soft, acidic waters more stringent criteria may be necessary, but for alkaline waters the criteria may be too stringent.

Standards from other countries

The US EPA (US EPA 1986) have established guidelines for chromium (VI) and (III) for the protection of fresh water organisms. Standards are based on four day and one hour averages which may be exceeded once every three years on average.

Values for chromium (VI) are set at 11 and 16 µg/l as four day and one hour averages respectively.

However, values for chromium (III) are calculated from formulae requiring a value for water hardness. The one hour average concentration (µg/l), which must on average not be exceeded more than once every three years is obtained from the formula;

$$e^{(0.8190[\ln(\text{hardness})]+3.688)}$$

The four day average concentration (µg/l), which must, on average, not be exceeded more than once every three years, is calculated from the formula;

$$e^{(0.8190[\ln(\text{hardness})]+1.561)}$$

The CCREM (1987) have set a guideline value for the protection of aquatic life stating that total chromium should not exceed 0.02 mg/l.

Recent data

Since Mance *et al* (1984b) there have been a number of studies on the effects of Cr VI to salmonids, however the majority are for the non-indigenous rainbow trout, these are outlined below.

In soft water (11 mg CaCO₃/l) Grande and Andersen 1983, found significant mortality (70%) of Atlantic salmon (*Salmo salar*) after chronic exposure to 0.1 mg Cr VI/l for 113 days. At 0.01 mg/l mortality was 10%. Also in soft water (25 mg CaCO₃/l), Stevens and Chapman (1984) found the NOEC for hatching of rainbow trout to be 0.048 mg Cr VI/l.

Van der Putte *et al* (1982), investigated for up to 32 weeks the toxicity of Cr VI to embryos, alevins and yearling rainbow trout (*Oncorhynchus mykiss*) in water of medium hardness (80 mg CaCO₃/l). Embryos were the most sensitive with significant mortality occurring at 0.2 mg/l. For all life stages no observed effects on survival and growth were seen at 0.02 mg/l.

Billard and Robaud (1985) found when eggs and sperm of rainbow trout were mixed in 0.005 mg/l Cr VI for 40 minutes there was a decrease in fertilisation. No details of water hardness were given.

Behavioural effects of exposure to Cr VI were noted by Anestis and Neufeld (1986). They reported the avoidance threshold of yearling rainbow trout to be 0.028 mg/l.

Toxicity data on the effects of Cr III are limited. Stevens and Chapman (1984), found in soft water (25 mg CaCO₃/l) increased hatch failure (15%) at 0.089 mg/l for fertilised eggs (rainbow trout) and 98% failure at 0.27 mg/l. For eyed embryos the highest NOEC for hatching was 0.16 mg/l, 70% hatch failure occurred at 0.27 mg/l. The limited data on the toxicity of Cr III to salmonids shows that it appears to have a similar toxicity to Cr VI at least for early life stages of trout.

Mance *et al* (1984b) compared environmental concentrations of chromium VI in UK rivers with the proposed EQS. It was unclear as to whether levels would reach the proposed EQSs for salmonids due to the high limits of detection used. However, where data on low concentrations was available, the annual average concentration did not exceed 2 µg/l.

Discussion

The new toxicity data have mainly been for chronic exposure, this indicates that the current EQS adopted by the DoE (1989) should adequately protect resident salmonid fisheries, at least older life stages. However, Billard and Robaud have indicated that a decrease in fertilisation for rainbow trout may occur at concentrations as low as 0.005 mg/l and therefore the current EQS may be insufficient for the protection of early life stages, further investigations need to be carried out in this area. The avoidance of rainbow trout at concentrations of 0.028 mg/l indicates that salmonids may actively avoid areas of chromium concentrations that may be acutely high but this would not be possible for early life stages. It is therefore proposed that the current EQSs for chromium while adequate for the protection of older life stages of salmonids, appear insufficient to protect early life stages. Because a

review of the EQS is currently being carried out, a separate EQS for early life stages is not proposed here, however, it should be noted that levels of chromium reaching the EQS in breeding/hatching areas may be detrimental to the fishery.

2.2.10 Copper

Standards applicable to the UK

Standards for copper (Cu) for the protection of fresh water fish are divided into two categories, EC-designated and non-EC-designated fisheries.

The EC (CEC 1978) has set guideline standards for dissolved copper (expressed as 95 percentile) for EC-designated salmonid fisheries (Table 2.11). The standards vary according to water hardness, with derogations allowed for natural enrichment.

Table 2.11 Guideline values for dissolved copper (95 percentile) (CEC 1978) for EC-designated fisheries

Water hardness (mg/l CaCO ₃)	Dissolved copper concentration (µg/l)
10	5
50	22
100	40
300	112

EQSs for dissolved copper, as an annual average for non-EC-designated fisheries, were proposed by Mance *et al* (1984c) and after consultation, adopted by the Department of the Environment (DoE 1989) for the protection of sensitive aquatic life (salmonid fish). As with the EC standards, the standards are set in relation to water hardness, Table 2.12. Mance *et al* (1984c) stated that in the presence of organic matter the acceptable concentration may be significantly higher.

Table 2.12 Environmental quality standards for the protection of salmonid fish (DoE 1989)

Water hardness (mg/l CaCO ₃)	Dissolved copper (annual average µg/l)
0 - 50	1
50 - 100	6
100 - 250	10
250+	28

The EQSs for copper for the protection of fresh water fish are currently being reviewed. Any new proposed standards will undergo a period of consultation in the Department of the Environment before being adopted.

EIFAC (Alabaster and Lloyd 1982) previously recommended tentative 50 percentile and 95 percentile standards, in relation to water hardness, for the protection of fresh water fish. The 95 percentile standards have since been adopted for use in EC-designated fisheries (CEC 1978). While those recommended as 50 percentile were recommended as EQSs by Mance *et al* (1984c) and have since been adopted for the protection of non-EC-designated fresh water fisheries (DoE 1989).

Standards from other countries

Standards for copper for the protection of aquatic life have been set by the US EPA (1986) and the CCREM (1987), these are outlined below.

The US EPA (1986) have set guidelines for the protection of all aquatic animals for one hour (instantaneous criteria always to be met) and four day average (chronic exposure) concentrations ($\mu\text{g/l}$), that may be exceeded no more than once every three years (the time believed to allow an unstressed system to recover from a pollution event in which copper exceeds the criteria). The calculations for the criteria are based on water hardness:

1. For four day average concentration ($\mu\text{g/l}$): $e^{(0.8545[\ln(\text{hardness})]-1.465)}$
2. For a one hour average ($\mu\text{g/l}$): $e^{(0.9422[\ln(\text{hardness})]-1.464)}$

Examples of the values obtained are given in Table 2.13

Table 2.13 Examples of copper criteria for the protection of aquatic animals (US EPA 1986)

Water hardness (mg CaCO_3)	1 day average concentration ($\mu\text{g/l}$)	4 day average concentration ($\mu\text{g/l}$)
50	9.2	6.5
100	18	12
200	34	21

The CCREM (1987) recommend that for the protection of all fresh water aquatic life values of total copper should not exceed those given in Table 2.14. The guideline values vary according to water hardness.

Table 2.14 Recommended guidelines for total copper for waters of different hardness (CCREM 1987)

Hardness (mg/l as CaCO ₃)	Concentration of copper (µg/l)
0 - 60	2
60 - 120	2
120 - 180	3
>180	4

Discussion

In assessing copper toxicity US EPA (1986) reviewed data on acute toxicity for species in 41 genera of fresh water organisms and chronic data from 15 fresh water species. Acute data showed toxicity in the range 16.74 µg/l to 10 240 µg/l at a water hardness of 50 mg/l. Data from some acute toxicity tests indicated that acute toxicity decreased as water hardness increased and that toxicity decreased with increasing alkalinity and total organic carbon. The criteria were developed to be based on an 'acid-soluble' method which they believe to be a more scientifically correct basis upon which to establish criteria for metal. However, since there are currently no EPA approved methods, they recommend the criteria to be applied to the total recoverable method.

The CCREM (1987) guidelines for total copper for the protection of fresh water aquatic life are based on two different criteria. For water hardnesses 0 - 60 mg CaCO₃/l the criteria are cited to be based on a guideline from Demayo and Taylor (1981). Criteria for water hardnesses 60 - >180 mg CaCO₃/l are based on the US EPA (1986) calculation for four day average concentration, for chronic toxicity, (using the lowest water hardness in the band calculated), the result was then multiplied by an application factor of 0.2, because they believed the effect of water hardness on chronic toxicity to be inconclusive.

No additional data for copper toxicity to that presented by EIFAC (Alabaster and Lloyd 1982) were reviewed in Mance *et al* (1984c). EIFAC could only propose tentative recommendations due to paucity of field observations. In hard water acutely lethal concentrations of copper to European species of fish varied up to one and a half orders of magnitude and there were no reliable data for comparison of species in soft waters. Toxicity in natural waters, with the exception of soft water free from organic matter and suspended solids, is less than predicted from laboratory tests, probably because copper is readily complexed by inorganic and organic substances and adsorbed onto particulate matter.

Reviewing data on the concentration of copper in UK rivers led Mance *et al* (1984c) to conclude that the majority of rivers easily complied with the EC guideline standards, however they mentioned that problems may occur in soft water regions.

Since the adopted EQSs for copper (DoE 1989) are currently being reviewed by WRc for the Department of the Environment, recent toxicity data has not been reviewed. The EQSs (DoE) currently adopted for the protection of salmonid fish are based on acute and chronic toxicity data, but there is a lack of field observation. They are similar to the four day average criteria (for chronic exposure) proposed by the US EPA (1986), which also reviewed chronic and acute toxicity data and used a formula based on water hardness for calculations, however, these also lack data on field observations. The CCREM (1987) are considerably lower than DoE (1989) and US EPA (1986) because they believed the effect of water hardness on chronic toxicity of copper to be inconclusive.

The adopted EQSs are similar to those proposed by the US EPA (1986) after a separate review, it therefore appears that they are adequate for the protection of salmonid fisheries, at least based on laboratory acute and chronic studies, but this cannot be concluded until the recent review is available. There is, however, a need for more information on field studies in order to ascertain levels toxic in the environment, where factors such as suspended solids and organic matter may affect toxicity.

2.2.11 Inorganic lead

Standards applicable to the UK

Environmental quality standards for annual average dissolved concentrations of lead (Pb) have been proposed by Brown *et al* (1984) for the protection of salmonids. These have since been adopted by the Department of the Environment (DoE 1989). The standards are related to water hardness and are given in Table 2.15. Brown *et al* (1984) stated that other values may be appropriate to local situations such as the protection of very sensitive species, or when organolead is present as a significant proportion of the dissolved lead. The EQSs are currently under review by WRc for the DoE.

Table 2.15 Environmental quality standards proposed by Brown *et al* (1984) for the protection of salmonids

Water hardness (mg/l CaCO ₃)	Annual average dissolved concentration of inorganic lead (µg/l)
<50	4
50 - 150	10
>150	20

Standards from other countries

Standards for the protection of fresh water aquatic organisms have been proposed by the US EPA (1986) and the CCREM (1987), these are relevant to the protections of salmonid fisheries and are outlined below.

The US EPA (1986) has set two guidelines for the protection of aquatic organisms, a four day average concentration and a one hour concentration (considered as an instantaneous concentration to be met at all times). The four day average concentration, that must not be exceeded more than once in every three years, is a limit given by a water hardness based calculation of;

$$e^{(1.273[\ln(\text{hardness})]-4.705)}$$

The one hour average concentration must not be exceeded, more than once in three years on average, the value is given by the water hardness based calculation;

$$e^{(1.273[\ln(\text{hardness})]-3.460)}$$

The EPA believes that measurements based on 'acid-soluble' concentrations would be more scientifically correct, however, in the absence of an US EPA approved method the criteria are based on total recoverable methods.

The CCREM (1987) have set guidelines that should not be exceeded, with respect to water hardness (Table 2.16). These are calculated using the US EPA (1986) calculation for four day average concentration (for chronic toxicity). However, they state that due to lack of toxicity data on lead in soft waters, a water hardness of 50 mg CaCO₃/l is used to calculate the guideline so as to afford adequate protection.

Table 2.16 Recommended guidelines for total lead (CCREM 1987)

Hardness (mg/l CaCO ₃)	Concentration of lead (µg/l)
0 - 60	1
60 - 120	2
120 - 180	4
>180	7

Recent data

Brown *et al* (1984) reviewed concentrations of inorganic lead in UK surface waters. Levels of inorganic lead were generally found to be low. However, exceptions may occur in rivers draining old mine workings and urban-industrial complexes where higher concentrations may occur during storm flows.

The majority of toxicity data available for salmonids in Brown *et al* (1984) were concerned with the non-indigenous rainbow trout. The data indicated that there was an increase in toxicity of dissolved lead with a decrease in water hardness. Also, based on these data, Brown *et al* (1984) found that an EQS for soft waters would be too stringent compared to field data on indigenous populations of brown trout, thus the evaluation for the EQS for soft water was based on field observations rather than laboratory tests.

Giattina and Garton (1983) investigated the avoidance behaviour of rainbow trout exposed to inorganic lead salts. In soft water (26 - 31 mg CaCO₃/l) trout avoided lead concentrations of approximately 26 µg Pb/l.

Grande and Anderson (1983) looked at the toxicity of lead to different developmental stages of Atlantic salmon (*Salmo salar*) exposed to lead nitrate in soft water (11 mg CaCO₃/l). For eggs, alevins, swim-up fry and eight month old salmon the maximum acceptable threshold concentration (MATC) based on survival was approximately 20 - 50 µg Pb/l.

Tang and Garside (1987) investigated the effect of pre-exposure to lead on the subsequent resistance of brook trout, *Salvelinus fontinalis*. Yearling trout were exposed to a range of concentrations of lead, 10 - 37 µg/l for 10 - 40 days and then exposed to 2 mg/l (normally lethal). The authors determined a MATC of 24 - 37 µg Pb/l during pre-exposure based on a decrease in resistance to lethal lead exposure. This is similar to the results reported by Grande and Anderson (1983).

Reader *et al* (1989) investigated the effect of pH on lead toxicity to yolk-sac brown trout fry (*Salmo trutta*) in artificially softened water. At pH 4.5, 19.2 µg Pb/l caused 95% mortality over 30 days, but at pH 6.5 mortality was less than 7% and in the control mortality at pH 6.5 was 0% but 10% at pH 4.5: two explanations were considered by the authors, sublethal stresses of H⁺ and metal toxicity may combine to give lethal effects; or metal toxicity may vary with pH as a result of metal speciation changes.

Haux *et al* (1986) in a field study investigated the physiological effects on juvenile whitefish, *Coregonus* sp., in soft (10 - 20 mg CaCO₃/l) lake water. Total lead concentrations of approximately 3 µg Pb/l caused a decrease in sodium plasma levels and 85% inhibition of erythrocyte d-ALA (aminolevulinic acid dehydrogenase). However, despite this there were no signs of anaemia.

Discussion

Toxicity data reported since Brown *et al* (1984) do not indicate lead toxicity to fish is significantly different than previously reported. In fact the avoidance behaviour noted by Giattina and Garton (1983) suggests that rainbow trout may be able to detect and avoid lethal concentrations of lead. Comparison of the US EPA (1986) guidelines and the EQS adopted by the DoE (1989) is difficult since one is for total lead and the other for dissolved lead. In addition, the database on which the hardness formulae were based is not given, and hence their relevance to indigenous salmonids cannot be assessed. It is therefore suggested that the standards currently adopted by the DoE (1989) for the protection of salmonid fish are adequate for the protection of resident populations.

2.2.12 Inorganic tin

Standards applicable to the UK

A tentative environmental quality standard as an annual average concentration for inorganic tin (Sn), of 25 µg/l total Sn has been proposed by Mance *et al* (1988b) for the protection of fresh water life, and this standard should protect salmonid fisheries. This has not yet been adopted by the Department of the Environment.

Standards from other countries

There do not appear to be any standards for inorganic tin proposed by other countries for the protection of salmonid fisheries.

Discussion

Mance *et al* (1988b) proposed a tentative EQS for the protection of all fresh water aquatic life (see above). However, this standard was based on a single study of the invertebrate *Gammarus pulex*, rather than on fish toxicity data. Available salmonid toxicity data were extremely limited, and only one study, Oshima (1931), on non-indigenous rainbow trout was reported. There do not appear to be any additional data on the toxicity of inorganic tin to salmonids.

Data on environmental concentrations are also very limited. Mance *et al* (1988b) only reported data on concentrations of inorganic tin from North American fresh waters. However, concentrations appeared to be limited to levels of nanograms/l.

What available data there are on the toxicity of inorganic tin to salmonids has been reviewed in Mance *et al* (1988b). For salmonids, these were found to be limited to only one study on a non-indigenous fish species and thus a separate standard for salmonids species could not be proposed. No additional data to that supplied in Mance *et al* (1988b) are available. It is therefore not possible, due to the lack of data, to recommend a separate EQS for salmonid fisheries. That previously proposed for all fresh water life appears adequate in view of the paucity of the data.

Before a standard for inorganic tin for salmonid fisheries can be proposed more information on acute and chronic effects on salmonids and levels in the environment needs to be obtained.

2.2.13 Iron

Standards applicable to the UK

Mance and Campbell (1988) proposed an environmental quality standard for total iron (Fe) of 2 mg Fe/l for the protection of fresh water fish. After consultation with the Department of the Environment, a standard of 1 mg/l as an annual average of dissolved iron was adopted (DoE 1989) for the protection of sensitive aquatic life (salmonids).

Standards from other countries

Water quality standards for the protection of fresh water aquatic life have been proposed by US EPA (1986) and the CCREM (1987).

The US EPA (1986) propose a standard of 1.0 mg/l total iron for the protection of fresh water aquatic life, however the derivation of this standard is not available. The CCREM (1987) recommend a much lower standard for the protection of aquatic life, that total iron should not exceed 0.3 mg/l.

Discussion

Mance and Campbell (1988) concluded that the available toxicity data did not show a difference in the toxicity of iron to salmonid or cyprinid fish and thus separate standards were not proposed. In addition, no variance in toxicity with respect to different water hardnesses was seen, however, below pH 7 the toxicity of iron increased and they stated that controlling authorities may need to reduce the EQS.

The EQS was based upon field data showing concentrations of total and dissolved iron in UK rivers at which a healthy fishery was sustained, rather than solely on laboratory toxicity data which the authors believed would be too restrictive. They stated EQSs based solely on the available laboratory toxicity tests would be in the range 0.3 mg Fe/l at pH 7 and 0.04 mg Fe/l at pH 6.5, considerably lower than levels found to support good fisheries.

Environmental data for salmonids cited in Mance and Campbell (1988) showed a healthy population of *Salmo trutta* (Scullion and Edwards 1980) to be present at 0.71 mg/l total iron. Geertz-Hansen and Matensen (1983), also for *Salmo trutta*, found mildly contaminated sites of 0.23 mg/l (dissolved) and 0.58 mg/l (total) iron, affected egg viability but not alevins, however, at another site, egg viability was normal at 2.01 mg/l (dissolved) and 3.02 mg/l (total) iron. The authors concluded that, from information on levels of iron and fishery status in UK rivers, a poor fishery or absence of fish would probably not be due to iron. Since the review by Mance and Campbell (1988) there appear to be no additional relevant data on the toxicity of iron to salmonids.

The CCREM (1987), when reviewing standards for total iron for the protection of fresh water aquatic organisms, cited Sykora *et al* (1972) who found a 50% reduction in the hatchability of fathead minnow eggs at an iron concentration of 1.5 mg/l. They concluded that the 1 mg Fe/l limit set by the US EPA (1986) would be too high and instead recommended a limit of 0.3 mg Fe/l previously set by the Ontario Ministry for the Environment and the International Joint Commission for Canada and the United States (IJC) for the Great Lakes, Canada.

The CCREM (1987) proposed a standard based on laboratory data which has previously been stated by Mance and Campbell (1988) to have been too restrictive. The EQS proposed by Mance and Campbell (1988) was based on a review of toxicity data and then adjusted with respect to field data and was thus based on observations of indigenous populations. No further relevant data on the toxicity of iron to salmonids are available. Therefore, it is concluded that the EQS of 1 mg/l dissolved iron as an annual average should be sufficient to protect salmonid fisheries.

2.2.14 Mothproofing substances

Standards applicable to the UK

Zabel *et al* (1988a) have proposed EQSs for the protection of fresh water fish (as 95 percentiles) for chlorophenylid (PCSDs) sulcofuron, flucofuron and permethrin of 0.05, 25.0, 1.0 and 0.01 µg/l respectively (as total active ingredient). These have been adopted by the DoE (1989) for the protection of sensitive (salmonid) aquatic life.

Discussion

Zabel *et al* (1988a) reviewed six chemicals used in mothproofing, EQSs for the protection of fresh water fish were proposed for four of these, see above, but they were unable to propose EQSs for cyfluthrin and hexahydropyrimidine (HHP). Further information since this review is currently limited and the reader is directed to Zabel *et al* (1988a) for more information on the derivation of these standards.

2.2.15 Nickel

Standards applicable to the UK

Mance and Yates (1984a) proposed an EQS of an annual average concentration of dissolved nickel (Ni) for the protection of fresh water fish with respect to water hardness (see Table 2.17). These values have since been adopted (DoE 1989) for the protection of sensitive aquatic life (e.g. salmonid). These standards are currently being reviewed by WRc for the DoE.

Table 2.17 Environmental quality standards for nickel (Mance and Yates 1984a, DoE 1989)

Water hardness (mg/l CaCO ₃)	Annual average concentration dissolved nickel (µg/l)
<50	50
50 - 100	100
100 - 200	150
>200	200

Standards from other countries

Standards for nickel for the protection of fresh water aquatic life have been proposed by the US EPA (1986) and CCREM (1987).

The US EPA (1986) have proposed a 24 hour average guideline for the protection of fresh water aquatic life based on a formula concerned with water hardness and a formula for a maximum value not to be exceeded also based on water hardness. The formula used for the calculation of the 24 hour average is;

$$e^{(0.76[\ln(\text{hardness})]+1.06)}$$

While the formula for the maximum value not to be exceeded is;

$$e^{(0.76[\ln(\text{hardness})]+4.02)}$$

Examples of the standards given by these formula are presented in Table 2.18.

Table 2.18 US EPA (1986) criteria for total nickel

Water hardness (mg/l CaCO ₃)	Value calculated for 24 hour average (µg/l)	Concentrations not to be exceeded (µg/l)
50	56	1100
100	96	1800
200	160	3100

The CCREM, (1987) have recommended levels of nickel, with respect to water hardness, that should not be exceeded in order to protect fresh water aquatic life. These are given in Table 2.19.

Table 2.19 Recommended guidelines for total nickel for the protection of aquatic life (CCREM 1987)

Hardness (mg/l as CaCO ₃)	Concentration of total nickel (µg/l)
0 - 60	25
60 - 120	65
120 - 180	110
>180	150

Discussion

For the EQS adopted by the DoE (1989) the majority of data were based on non-indigenous fish, mostly rainbow trout. Three standards were proposed with respect to water hardness, those for soft and hard waters were based on experimental data on non-indigenous fish, while that for intermediate water hardness was calculated by interpolation. Data on environmental concentrations showed that there appeared no basis to alter the standards based on laboratory data, and concentrations of total nickel (particulate and dissolved) were not found to approach the EQS, and therefore the authors felt no reason to identify nickel as a limiting factor in rivers for fisheries.

The CCREM (1987) guidelines adopted a guideline recommended by Taylor, Demayo and Reeder (1979) for soft water (0 - 60 mg CaCO₃) and used the 24 average hardness related formula from the US EPA for the other standards. In addition, they cite US EPA (1980) stating that they had compiled data on acute toxicity to 22 species of aquatic animals. Those for acute toxicity to fish ranged upwards from 2.48 mg/l for rock bass (*Ambloplites rupestris*). Only data from one acute toxicity test for salmonids, rainbow trout, were found and these gave a value of 35.5 mg/l at unknown hardness (Hale 1977).

The hardness related formula for nickel toxicity used by the US EPA (1986) is based on a relationship between water hardness and acute toxicity of three species, two invertebrates and one fish, *Daphnia magna*, *D. pulicaria* and fathead minnow.

There have been a few relevant studies on salmonids since Mance and Yates (1984a). Nebeker *et al* (1985) investigated toxicity of nickel to various life-stages of rainbow trout in soft water (27 - 53 mg CaCO₃/l). Three and twelve month 96 hour LC50s were found to be 10 - 10.9 and 8.1 - 8.9 mg Ni/l respectively. In long term exposures on pre-swim up larvae and eyed eggs, the highest no effect concentration was 0.13 mg/l. In addition, 75 - 85 day exposures showed significant weight reduction at 0.06 and 0.04 mg/l respectively. Also in soft water (11 mg/l CaCO₃), Grande and Andersen (1983) found at 0.05 mg Ni/l, the hatching success of Atlantic salmon eggs was reduced.

In water of medium hardness, (104 mg CaCO₃), Birge (1978) found adverse effects on the embryo and larval development of rainbow trout at 0.05 mg Ni/l.

The data on nickel toxicity to fresh water fish were found by Mance and Yates to be limited, with the EQS for intermediate water hardness not being based on toxicological data. However, these standards are currently under review.

As previously stated, the US EPA (1986) standards are based on a water hardness formula. The standards proposed for total nickel as 24 hour average criteria are similar to those adopted for the DoE (1989) for dissolved nickel, however, those stated as concentrations not to be exceeded are an order of magnitude higher than those proposed by the DoE (1989). In addition, since the criteria are based on a formula related to water hardness, which in turn is based on a relationship only defined for three species (only one of which is a fish) the integrity of this formula must be called into question, and a larger data base should be constructed before such assumptions can be adequately assessed. This argument also applies for the criteria proposed by the CCREM (1987) which also use these formulae.

The data available since Mance and Yates (1984a) for soft water, indicate that the chronic early life stage tests are a factor of five higher than the EQS, and thus such stages should be adequately protected by this standard.

For water of intermediate hardness, Birge (1978) gave an EC50 of 0.05 mg/l for rainbow trout embryo and larval development which is lower than the current EQS of 0.1 mg/l. However, other work carried out by Birge and coworkers has been discredited.

It is therefore proposed that since a review of the EQS for nickel for the protection of fresh water fish is currently being carried out, the EQS currently adopted should be maintained, until any new proposals are given. However, it should be noted that in water of intermediate hardness some detrimental effects may occur if the current EQS is just met. Therefore, it may be appropriate to apply a more stringent standard, especially if early life stages are present.

2.2.16 Organotins

Standards applicable to the UK

Zabel *et al* (1988b) recently reviewed the available data on organotins and proposed EQSs of 20 ng/l (total) as 95 percentiles for tributyltin (TBT) and triphenyltin (TPT), for the protection of fresh water fish. These standards are applicable to salmonid fisheries.

The DoE (1989) have since adopted EQSs of 20 ng/l as a maximum allowable total concentration for TBT and TPT for the protection of sensitive (e.g. salmonid) aquatic life.

Discussion

In their review, Zabel *et al* (1988b) considered tributyltins, triphenyltins and tricyclohexyltins, however data for tricyclohexyltins were limited to an acute test on large mouth bass and thus an EQS could not be proposed. Data for TBTs and TPTs were limited, with no field data available for either and thus Zabel *et al* (1988b) considered that the proposed EQSs could only be considered as tentative. For TPT, Zabel *et al* (1988b) found a lack of available chronic data but since the acute data for fish and invertebrates were similar to that for TBT they proposed a similar EQS.

The current data on organotins and their toxicity to salmonids are limited, and further field and chronic data, which are currently unavailable, are needed before standards specifically for the protection of salmonid fisheries can be proposed. It can only therefore be considered that the current tentative EQS adopted by the DoE (1989) will adequately protect salmonid fisheries.

2.2.17 pH

Standards applicable to the UK

The EC, (CEC 1978) in the Fisheries Directive have set a mandatory level of pH 6 - 9 as a 95 percentile for the protection of EC-designated salmonid fisheries. Sampling frequency is stated to be monthly (which may be reduced if water is known to be well within the specified limits, where pollution is absent and where there is no risk in the deterioration of water quality). The range of pH is based on the assumption that concentrations of other potentially harmful substances are low. The Directive also sets a condition that artificial variations of pH with respect to background levels should not exceed ± 0.5 pH units within the mandatory levels and that such changes should not increase the harmfulness of other substances in the water. Derogation is allowed in the event of exceptional weather, geographical conditions or natural enrichment.

Wolff *et al* (1988) have recently carried out a major review on the effects of pH on fisheries. They proposed an EQS for the protection of fresh water fish of pH 6.0 - 9.0 also as a 95 percentile (monthly sampling). This is the same as set by the EC (CEC 1978). This standard has since been adopted by the DoE (1989) for the protection of non-EC-designated salmonid fisheries.

Standards from other countries

Standards for pH for the protection of aquatic life have been set for Canada and by the US EPA.

The US EPA (1986) concluded that the pH range 6.5 - 9.0 would protect fresh water aquatic life. The same range is set as a guideline by the CCREM (1987) and is based on the review by EIFAC (Alabaster and Lloyd 1982) and the US EPA criteria.

Recent studies

When proposing the EQS, Wolff *et al* (1988) found no direct correlation between pH values and fishery status and thus derived the EQS from subjective assessment rather than the application of an arbitrary safety factor to results from acute toxicity tests.

There have been a large number of reviews carried out on the effects of pH on fresh water fisheries and these are mentioned in Wolff *et al* (1988) who, as previously mentioned, carried out the most recent review.

One of these reviews that is of relevance to UK fresh water fisheries was by EIFAC (Alabaster and Lloyd 1982). They reviewed the available data on the effects of different pH levels to European fresh water fish. It was concluded that the pH range 6.5 - 9.0 would be harmless to any species of fish, but recognised that a wider range could be tolerated by many species such as salmon (pH 5.3 - 9), cyprinids (pH 4.5 - 9.5) and pike (pH 4 - 9.5). They stated that there was no definite pH range within which a fishery was unharmed but rather a gradual deterioration would occur as the pH values became further removed from the normal value. The pH range 5 - 9 was concluded not be directly toxic to fish, but the toxicity of several pollutants may be markedly changed within this range.

Wolff *et al* (1988) reviewed factors influencing pH and the chemistry affected by pH, including buffering capacity by bicarbonate, ammonia equilibrium, dissolution and speciation of metals, which in turn may have toxic effects on fish. Aluminium was regarded as a particular problem, its solubility being greatest below pH 4 and above pH 9, but with mobilisation increasing as the pH goes below 6. The toxicity of aluminium has been reviewed by O'Donnell *et al* (1984).

Environmental concentrations for UK rivers were found by Wolff *et al* (1988) to be in the mean pH range 6 - 9. However, lower pHs were found in rivers and lakes in regions of very low water hardness, while high pH values were found in waters associated with high algal blooms.

The majority of toxicity data in Wolff *et al* (1988) were for low pHs. Summarising toxicity tests showed that mortality for adults occurred below pH 5, for fry and embryos below pH 6.0, reduced growth occurred below pH 6, while stress to fish was reported in the range pH 4.0 - 6.0 and above pH 9.8. Field data indicated that the absence of fish was not necessarily solely due to pH, with other factors, especially trace metal, high aluminium and low calcium concentrations believed to increase pH toxicity to fish.

Watt (1987) investigated the effect of acidification on Atlantic salmon using angling catch data from 1936 to 1985 for 18 rivers similar in state to 50 years previous, in Nova Scotia, Canada. For rivers less than pH 4.7, angling records ended in the early 1970s and electrofishing failed to find any surviving juvenile salmon. The authors concluded salmon runs no longer exist. For rivers in the range pH 4.7 to 5.0 the angling catch had declined to approximately 10% of 1936 to 1953 levels. Electrofishing indicated the presence of salmon juveniles, usually in higher pH tributaries, these were believed to function as natural refuges. The rivers in the higher two pH ranges, (i.e. above pH 5.1) showed no signs of an impact of acidification on angling returns, with the exception of the lower pH tributaries of rivers in the pH 5.1 - 5.4 range, here electrofishing indicated acidification limited juvenile salmon production.

A number of studies have concluded that the decline of a fishery is due to a gradual recruitment failure due to episodic acidification rather than massive fish kills. These acid episodes, (flushes of acid water) are most common during periods of heavy rain and rapid runoff when there is little opportunity for the buffering effect of any mineral soil to occur (MAFF 1991). Hesthagen (1989) investigated fish kills of Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) associated with environmental acidification during snowmelts in south western Norway. Fish kills were found to be episodic, the mortality varying considerably between the years investigated, usually following sudden increases in discharge and depressions in river pH. The authors concluded that the disappearance of Atlantic salmon and brown trout stocks were not usually in the form of massive fish kills but rather through recruitment failure. A predominant fraction of the dead fish were large parr which were presumed to be potential smolts that year. In addition, Turnpenny *et al* (1988) investigated the factors limiting fish populations in Loch Fleet, an oligotrophic upland lake in Galloway, south west Scotland which had supported a population of brown trout (*Salmo trutta*). The authors concluded that the acidity combined with low calcium and high aluminium were the main factors that had affected the fish population. However, while the decline in the fish population started in 1950, acute acidification did not occur until 1975. The authors attributed the decline over 1950 - 1975 to be probably due to periodic acidification of the spawning streams which resulted in recruitment failure.

Discussion

The available data indicate that declines of fisheries are usually due to a series of episodic events of decreased pH that result in a gradual recruitment failure, rather than one massive fish kill. Because such events are usually due to periods of increased rainfall, it is not possible to set standards that will protect against such effects. However, standards, that will protect against direct toxicity, or the indirect effects of changes in toxicity of elements due to change in the pH in the long term due to discharges, are necessary for the protection of resident populations. The reviewed data, while mainly on effects of low pH rather than high, indicate that the present EQS for the protection of salmonid fisheries, pH 6 - 9 (as a 95 percentile on an annual basis) (DoE 1989) is adequate for the protection of resident salmonid fisheries.

2.2.18 Sulphide

Standards applicable to the UK

Mance *et al* (1988c) proposed an EQS for sulphide for the protection of fresh water fish as undissociated hydrogen sulphide. The EQS (Table 2.20) is expressed with temperature and oxygen ranges and as an annual average concentration and a 24 hour maximum average concentration, which if exceeded could cause detrimental affects.

Table 2.20 EQS for sulphide, expressed as undissociated hydrogen sulphide ($\mu\text{g/l}$) (Mance *et al* 1988c)

Temperature and oxygen range	Annual average concentration ($\mu\text{g/l}$)	24 hour maximum average concentration ($\mu\text{g/l}$)
<15°C <5 mg O ₂ /l	0.5	5.0
<15°C >5 mg O ₂ /l	1.0	10.0
>15°C <5 mg O ₂ /l	0.25	2.5
>15°C >5 mg O ₂ /l	0.5	5.0

The DoE has not yet adopted any standards for sulphide for the protection of fresh water fish. WRc is currently undertaking a review of sulphide and this EQS.

Discussion

Mance *et al* (1988c) found that information on the toxicity of sulphides to fresh water fish was largely restricted to studies of hydrogen sulphide. Little variation in the sensitivity of the fish tested was found, and the toxicity of hydrogen sulphide was not found to increase dramatically with an increase in the duration of exposure. Because of the effect of temperature and oxygen concentration on the toxicity the EQS was proposed as concentrations for particular temperature or oxygen ranges.

Since a review of sulphide is currently being undertaken by WRc for the DoE further data are not presented here. From the data presented in Mance *et al* (1988c) the EQS should be adequate to protect salmonid fisheries.

2.2.19 Vanadium

Standards applicable to the UK

Tentative EQSs, set with respect to water hardness, were proposed by Mance *et al* (1988d) for total vanadium (V) for the protection of fresh water fish, these are given in Table 2.21. The standards have since been adopted by the Department of the Environment (DoE 1989) for the protection of sensitive aquatic life (e.g. salmonids).

Table 2.21 Environmental quality standards for total vanadium for the protection of fresh water fish (Mance *et al* 1988d, DoE 1989)

Water hardness (mg/l CaCO ₃)	Annual average concentration (µg V/l)
< 200	20
> 200	60

Standards from other countries

There appear to be no standards for the protection of fresh water fish or more specifically salmonid fish set by other countries.

Discussion

Reviewing the available data, Mance *et al* (1988d) found toxicity data for salmonids were limited to the non-indigenous fish, rainbow trout. The majority of data available for the derivation of this standard were short term LC50s, reported as nominal concentrations, and thus are likely to be an overestimation of the true exposure concentration. The limited toxicity data indicated that toxicity to fresh water fish appeared to decrease with increasing water hardness, and hence standards were proposed for water hardness of greater and less than 200 mg CaCO₃/l.

A review of available data since Mance *et al* (1988d) has shown it to be very limited. Hamilton and Buhl (1990) reported 24 and 96 hour LC50s of 39.2 mg/l and 16.5 mg/l for chinook salmon in reconstituted fresh water of hardnesses 211 mg CaCO₃/l and 343 mg CaCO₃/l respectively (pH 7.4 - 8.3). In addition, Hamilton and Buhl (1990) cited previous work on brook trout (*Salvelinus fontinalis*) which they reported to show similar toxicity. These values fall within the range of toxicities reported by Mance *et al* (1988d).

Field data reported by Mance *et al* (1988d) from Northumbrian WA region indicated that at the highest concentrations reported in rivers, 20 and 40 µg/l (water hardness not stated), both supported good sport fisheries.

New data, since Mance *et al* (1988), on the toxicity of vanadium to salmonids are limited to two studies. These indicate that the EQSs for protection of fresh water fish, at least at the higher hardness value, would give adequate protection. It can be concluded that the current EQSs (DoE 1989) for the protection of fresh water fish are adequate for the protection of salmonid fisheries.

2.2.20 Zinc

Standards applicable to the UK

Standards in the UK for zinc have been set by the EC for EC-designated salmonids fisheries, and by the DoE (1989) for the protection of salmonids in non-EC-designated fisheries.

Standards for EC-designated salmonid fisheries have been set by the EC (CEC 1978) these are given in Table 2.22. The standards are mandatory and expressed as 95 percentiles of total zinc with a minimum sampling frequency of one sample per month. A derogation in the Directive allows the standards to be exceeded if caused by natural enrichment e.g. discharges from old mine workings.

Table 2.22 EC mandatory values for designated salmonid fisheries (CEC 1978)

Water hardness (mg/l CaCO ₃)	Total zinc (µg/l) 95 percentile
10	30
50	200
100	300
500	500

Environmental quality standards for total zinc for the protection of salmonids for non-EC-designated salmonid fisheries were proposed by Mance and Yates (1984b) and have since been adopted by the DoE (1989) for the protection of sensitive (salmonid) species. The EQSs are annual averages varying with respect to water hardness (Table 2.23). They are the same as those originally proposed by EIFAC (Alabaster and Lloyd 1982). The EQSs are currently under review by WRc for the Department of the Environment.

Table 2.23 EQSs for total zinc for the protection of salmonids (Mance and Yates 1984b, DoE 1989)

Water hardness (mg/l CaCO ₃)	Total zinc (µg/l) (annual average)
10	8
50	50
100	75
500	125

Standards from other countries

The US EPA (1987) have set two criteria for zinc, both of which are calculated from formulae that are based on water hardness.

The four day average concentration ($\mu\text{g/l}$) should not exceed, on average, more than once every three years, the value calculated from the equation;

$$e^{(0.8473[\ln(\text{hardness})]+0.7614)}$$

The one hour criteria (regarded as instantaneous) should not exceed, on average, more than once every three years, the value calculated from the equation;

$$e^{(0.8473[\ln(\text{hardness})]+0.8604)}$$

Examples of calculated values are given in Table 2.24.

Table 2.24 Calculated values of US EPA (1987) 4 day average and one hour zinc criteria. Values should not be exceeded more than once on average every three years

Water hardness (mg/l CaCO_3)	One hour average concentration ($\mu\text{g/l}$)	Four day average concentration ($\mu\text{g/l}$)
10	17	15
50	65	59
100	120	110
200	210	190

The CCREM (1987) gives a tentative guideline of zinc for the protection of fresh water aquatic life. The guideline states that the concentration of total zinc should not exceed 0.03 mg/l. The standard does not vary with water hardness since they believed there were insufficient data to show that chronic toxicity decreases as water hardness increases.

Recent data

Since Mance and Yates (1984b) a number of studies have reported zinc toxicity to salmonids, some of these are outlined below.

Bradley and Sprague (1985) reported for juvenile rainbow trout a 96 - 120 hour LC50 of 0.17 mg/l in soft waters at pH 7. Zinc was found to be most toxic at this pH. This corresponded to work by Cusimano *et al* (1986) who found the toxicity of zinc to be reduced at pH less than 7. The lowest acute toxicity value reported by them for rainbow trout was a 96 hour LC50 of 0.066 mg/l at pH 7 in soft water (9.2 mg CaCO_3/l). This is an order of magnitude lower than that reported by Bradley and Sprague (1985) but similar to a 96 hour

LC50 of 0.084 mg/l (20 - 22 mg CaCO₃/l) for chinook salmon (*Oncorhynchus tshawytscha*), reported by Finlayson and Verrue (1982). Anadu *et al* (1989) found 96 hour LC50s of 0.1 - 0.14 mg/l for juvenile rainbow trout in soft waters at pH 6.6. Acclimatised fish were less sensitive.

Everall *et al* (1989) looked at the sensitivity of brown trout (*Salmo trutta*) to zinc. Zinc again was most toxic at pH 7. During a 30 day experiment no effect on survival of fry occurred at 0.021 mg/l, pH 6.5 in soft water.

The CCREM (1987) recommended guideline of 0.03 mg/l total zinc was stated to coincide with measured 'no effect' concentration for rainbow trout and fathead minnows (0.036 and 0.03 mg/l respectively).

Discussion

The toxic effects of zinc to fish appear to be associated with water hardness, increased toxicity occurring in soft waters. There is also some evidence that zinc is more toxic around pH 7 than in more acidic or alkaline waters.

New data since Mance and Yates (1984b) have shown in soft water zinc is acutely toxic to rainbow trout at concentrations as low as 0.066 mg/l. Reader *et al* (1989) found significant mortality of *Salmo trutta* at 0.018 mg/l, but this was in acidic waters, (pH 4.5), and thus may not be suitable to be related to other waters. Only one test, Everall *et al* (1989), studied chronic exposure, with no effect after 30 days to the survival of brown trout fry at 0.021 mg Zn/l in soft water.

The US EPA (1987) one hour and four day criteria for soft waters, e.g. 10 mg CaCO₃/l (0.015 and 0.017 mg Zn/l respectively) appear to be adequate for the protection of salmonids fisheries, however, that proposed by the CCREM (1987), (0.03 mg Zn/l), according to toxicity data of Cusimano *et al* (1986) (96 hour LC50 of 0.066 mg/l for rainbow trout), may be too high to give adequate protection in soft waters.

The new data reported have been limited to toxicity tests in soft water, with no available additional data on the effects of zinc to salmonids in harder waters. However, the data indicate that, at least for soft waters, the EQS proposed by Mance and Yates (1984b) should adequately protect salmonid fisheries in soft waters. In the absence of additional data, it is also proposed that the standards proposed by Mance and Yates for harder waters should adequately protect salmonid fisheries.

2.3 Physical parameters

2.3.1 Temperature

The tolerance of fresh water fish to high temperature depends on a number of factors including the species, stage of development, season, acclimation temperature, concentrations of dissolved oxygen and the presence of pollutants. Standards for temperature for the protection of salmonid fisheries are necessary since changes in the natural régime may produce changes in the population composition, and fish behaviour.

Standards applicable to the UK

Standards for temperature for the protection of salmonid fisheries have been set by the EC (CEC 1978) for the protection of EC-designated salmonid fisheries, but none have been adopted for non-EC-designated fisheries.

The Directive on Fishing Water Quality (CEC 1978) gives three mandatory temperatures which apply to designated salmonid fisheries. All are concerned with discharges and are expressed as 98 percentiles (i.e. can be exceeded for 2% of the time). The first states that the temperature measured downstream of a point of discharge at the edge of a mixing zone must not exceed the unaffected temperature by more than 1.5°C. Derogations that are limited in geographical scope may be allowed for conditions where the competent authority can prove that there are no harmful consequences for the balanced development of the fish population. For the second and third values, the Directive states that thermal discharges must not cause the temperature downstream of the point of discharge (at the edge of the mixing zone) to exceed 21.5°C or 10°C in waters used by cold water breeding species, during the breeding period.

Standards for temperature, which are suitable for protection of UK non-EC-designated fisheries have been proposed by EIFAC (Alabaster and Lloyd 1982) and the Anglian Water Authority (1986), these are outlined below.

EIFAC (Alabaster and Lloyd 1982) reviewed the effect of temperature on European inland fisheries and tentatively recommended guidelines that are divided into three seasons which should be applied to waters not polluted by chemicals:

1. Autumn and Winter: Raising the temperature by 5 - 6°C above norm in winter and autumn may result in higher mortality among embryos of salmonids and may reduce the size, weight and viability of hatched fry. An increase in the temperature of 2°C above the norm in autumn and winter at the time of reproduction of whitefish (*Coregonus sp.*) would be damaging to the reproduction of this species although the majority of other species would be unaffected.
2. Spring: For most spring-spawning fish the optimum range of temperature for spawning and embryonic development is no greater than 8°C.
3. Summer: For *Salmo spp.* an upper permissible level of 20 - 21°C was recommended, while for Coregonids (with the exception of embryos) since they are able to withstand a 5 - 6°C rise in temperature, a maximum upper limit of 22 - 23°C was recommended.

Anglian Water Authority (1986) have also proposed water quality criteria for temperature in salmonid dominated rivers. These are a 95 percentile of 20°C and a 99 percentile of 22°C.

Standards from other countries

The US EPA (1986) have criteria for the protection of all fresh water aquatic life. The criteria consist of two upper limiting temperatures for any given location which are based on the important sensitive species found there at that time. These standards have also been adopted for use in Canada (CCREM 1987).

One limit is the maximum temperature for short exposure, this is time dependent and given by a species specific equation. The second limit is a maximum average weekly temperature (MAWT) that may be based on one of four criteria:

1. In cooler months, the MAWT is an elevated temperature that will protect against mortality of important species if the elevated plume temperature is suddenly dropped to the ambient temperature, the limit being equal to the acclimation temperature minus 2°C when the lower lethal threshold temperature equals the ambient water temperature.
2. In the warmer months, the MAWT is determined by adding the physiological optimum temperature (usually for growth) and a factor calculated as one third of the difference between the ultimate upper incipient lethal temperature and the optimum temperature for the most sensitive important species normally found at that location at that time.
3. In the reproductive season, the MAWT is the temperature that meets site specific requirements for successful migration, spawning, egg incubation, fry rearing and other reproductive function of important species.
4. At a specific site, the MAWT is that found necessary to preserve normal species diversity or prevent the appearance of nuisance organisms.

Calculated examples for salmonids, of short-term temperature maxima for survival for juveniles and adults during the summer, maximum weekly average temperatures for growth (b) along with reported optimal temperature for spawning and upper temperature for successful incubation and hatching (ERL-Duluth 1976, reported in US EPA 1986) are given in Table 2.25.

Table 2.25 Calculated examples of US EPA (1986) temperature criteria

Species	Short term maxima °C	MAWT (Growth)(b) °C	Reported optimum temperature °C spawning	Reported optimum temperature °C incubation/ hatching
Atlantic salmon	23	20	5	7
Brook trout	24	19	9	13
Coho salmon	24	18	10	13
Rainbow trout	24	19	9	13
Sockeye salmon	22	18	10	13

Recent data

A large number of studies on the effect of temperature on salmonid fisheries are available, these have been reviewed and salient articles are outlined below.

Early-life stages

Changes in the temperature of redds, such as may occur during dewatering of salmon redds due to fluctuations in water flow may effect the intergravel development of salmonids. Becker *et al* 1982, (cited in Neitzel and Becker 1985), noted intergravel temperatures rose from 10°C to 28°C during 16 hour dewatering.

Effects of such temperature rises on four intergravel development stages, cleavage eggs, embryos, eleutheroembryos and pre-emergent alevins of chinook salmon, were investigated by Neitzel and Becker (1985). Organisms were acclimated to 10°C and subjected to heat shock tests involving exposure to increased temperatures for up to eight hours. The ability to tolerate heat shock exposure was found to vary with developmental stage but few deaths in any stage occurred at 22°C (water temperature) or below. Cleavage eggs were the least tolerant, and embryos were the most tolerant surviving eight hour exposures to 25°C and two hour exposure to 26.5°C.

Humpesch (1985) determined optimum temperatures for embryonic development and hatching of winter and spring spawners. For winter spawners, *Salvelinus alpinus*, *S. fontinalis* and *Salmo trutta*, temperature ranges were <1.4 to <15°C and 1.4 - 8°C for embryonic development and hatching respectively, while for for spring spawners, *Oncorhynchus mykiss*, *Hucho hucho* and *Thymallus thymallus*, they were found to be <5.0 to <19.0°C and 7 - 12°C. Investigations on time of development and temperature showed slower development at lower temperatures with the period between fertilisation and 50% hatched decreasing from ten weeks at 5°C to approximately four weeks at 15°C. This shows changes in the temperature of spawning sites could have an effect on the time of hatching, and thus survival.

Peterson *et al* 1977 (cited in Crisp 1987) reported salmonid eggs can tolerate a temperature range of 0 to 16°C although variations between species can be expected.

Warm and cold water discharges may affect the development of fish, especially if in the area of spawning grounds.

Morrison (1989) studied the effect of warm cooling water effluent from a distillery on the River Fiddic, Scotland, a spawning river for Atlantic salmon and brown trout, where the thermal discharge was found to raise temperature of the stream by 1 - 3°C above ambient all year round. Sites affected by the warm water discharge were found to result in earlier hatching of eggs of Atlantic salmon and brown trout and a faster rate of development, thought to be due to increased food availability, than areas upstream of the discharge.

In contrast, Salveit (1990) investigated the growth of Atlantic salmon and brown trout in a Norwegian river, affected by a cool water release from a power station compared to fish upstream of this discharge. Temperature data for the years 1987 and 1988 (July to August) showed that below the power station the water temperature was stable at c. 12°C in 1987 and in 1988 it did not exceed 14°C. Above the power station it reached 15°C in 1987 while in 1988 the temperature exceeded 20°C on a number of occasions. Slower growth rates for the Atlantic salmon and brown trout affected by the discharge were seen, with the result that smoltification occurred a year later than for fish in the upper parts of the river. The authors

suggested that this late smoltification would lead to high mortality due to increased predation, etc. in the extra year and therefore lower production of both adult salmon and brown trout.

Thomas *et al* (1986) investigated the effect of fluctuations in daily temperature on physiology and growth of two age classes (age 0 and age II pre-smolts) of juvenile coho salmon, *Oncorhynchus kisutch*. Fluctuations in daily temperature may occur for a variety of reasons, such as a reduction in cover, or decrease in flow and effects may be especially seen in the summer months when high fluctuations in daily temperatures may occur. Thomas *et al* (1986) found all fish survived the diel temperature range 5 - 23°C, and mortality was first noted at the 4 - 25°C range. The authors stated that juvenile coho salmon tolerated the high temperature phases of the cyclic temperatures about as well as they tolerated constant high temperatures in a study carried out by Brett (1952).

MAFF (1991) reported that salmon have been observed to spawn at temperatures of 0 - 8°C. The preferred temperature for the growth and survival of newly-hatched fish was reported to be 14 - 15°C while the preferred range for parr is 9 - 17°C. The absolute lethal temperature for young salmon is around 27°C but temperatures over 20°C will reduce growth and may create other problems such as increasing the chance of infection by diseases.

Crisp (1987) suggested that temperature may be an important trigger to salmonids, they cited Allen (1940) and Saunders and Henderson (1969) who found that at temperatures below 7°C Atlantic salmon parr do not feed well and Siginevich (1967) who found that at this temperature they show little growth.

Wilson *et al* (1987) reviewed the available literature to investigate the effect of temperature on five species of salmon; *Oncorhynchus keta* (chum), *O. kisutch* (coho), *O. gorbuscha* (pink), *O. nerka* (sockeye), *O. tshawytscha* (chinook), commonly found in the Susitna drainage, Alaska. Using data from the literature as well as observed effects in the Susitna drainage they estimated tolerance and preferred criteria for different life-stages for the five salmonid species (see Table 2.26). Tolerance ranges for spawning were estimated to be 1.0 to 18°C (the least tolerant being *O. gorbuscha* with a tolerance range of just 7.0 - 18°C), the preferred range for spawning was estimated to be 6.0 - 13°C. For incubation, the tolerance range for these five species was estimated to be 0 - 16°C, and the preferred range estimated to be 2 - 12°C.

Table 2.26 Temperature (°C) ranges for different life-stages of five salmonids (Wilson *et al* 1987)

Species/life-stage	Range from cited literature	Estimated tolerance	Estimated preferred
<i>Oncorhynchus keta</i>			
spawning	2.0 - 17.0	1.0 - 14.0	6.0 - 13.0
incubation	1.3 - 14.0	0.0 - 12.0	2.0 - 8.0
<i>O. nerka</i>			
spawning	4.9 - 13.0	4.0 - 14.0	6.0 - 12.0
incubation	2.0 - 14.3	0.0 - 14.0	4.5 - 8.0
<i>O. gorbuscha</i>			
spawning	7.0 - 18.4	7.0 - 18.0	8.0 - 13.0
incubation	0.5 - 15.0	0.0 - 13.0	4.0 - 10.0
<i>O. tshawytscha</i>			
spawning	5.6 - 13.6	5.0 - 14.0	7.0 - 12.0
incubation	nr	0.0 - 16.0	4.0 - 12.0
<i>O. kisutch</i>			
spawning	2.0 - 17.0	2.0 - 18.0	6.0 - 11.0
incubation	1.3 - 14.0	0.0 - 14.0	4.0 - 10.0

Note: nr = not reported

Adults

Kilgour *et al* (1985) reported observed upper incipient lethal temperature (UILT) (the temperature at which 50% of the population survives an indefinitely long exposure, statistically analogous to LC50) for *Salvelinus fontinalis* (brook trout) (for acclimation temperatures 11 to 24°C) of 23.8 to 25.3°C. For *Oncorhynchus spp.* observed tilts were in the range 21.3 to 25.1°C (for acclimation temperatures 5 to 24°C).

Discussion

The reviewed literature has indicated various optimum temperature ranges for different life cycle stages for salmonids, mainly non-indigenous. It would be impossible to set standards for each life cycle stage, due to differences in species in rivers and their preferred temperature range. Instead a temperature standard above which a salmonid fishery may be adversely affected would be better. The data indicate that both adults, and at least for a limited period, early life stages will survive if the temperature does not exceed 20°C. This value is similar to that previously set/proposed for various water quality standards. The EC (CEC 1978) state that salmonid fisheries will be protected at temperatures below 21.5°C,

EIFAC (Alabaster and Lloyd 1982) state the upper permissible level for *Salmo spp.* in summer is 20 to 21°C and the Anglian Water authority have guidelines in salmonids dominated rivers of 95 percentile at 20°C and 99 percentile at 22°C.

Based on the reported data (and in particular MAFF 1991), a maximum temperature of 20°C is proposed for the protection of salmonid fisheries. This maximum should only apply in waters affected by thermal discharges: natural temperature may exceed these values. This standard should adequately protect adult salmonids, however, while the data have indicated that early life stages may be affected at temperatures below 20°C, these stages are unlikely to be present at the times (most likely summer) when this temperature will be reached, additionally, since they are present in the bottom of streams/river, they are unlikely often to be subjected to large temperature variations.

In addition, to the proposed standard for maximum temperature, a standard for maximum temperature change due to thermal discharges is also proposed. The standard is for a temperature difference (up or down) from unaffected. Temperature differences of as little as 2 to 3°C may have an effect of delaying or accelerated development. Therefore, it is proposed that the EC Fish Directive standard for temperature difference be applied to non-designated waters also. That is, the temperature difference at the edge of the mixing zone, compared to unaffected water, should not exceed 1.5°C. This could be expressed as a maximum allowable difference which would be a more practicable expression than the 98 percentile in the Directive.

2.3.2 Suspended solids and turbidity

Increased suspended solids (SS) may cause a number effects on fish. These may be indirect or direct. Indirect effects include blanketing of the stream bed, which may affect the availability of food or blocking of gravel which may hinder spawning or cause a reduction in the flow of oxygenated water which may in turn affect the survival and development of eggs or parr. Direct effects include siltation of gills.

Certain factors may increase the concentration of SS. These include, afforestation, during the early stages of the forestry cycle and felling, and gravel extraction, which may cause the release of fine sediments.

Turbidity is an optical property of water whereby suspended and dissolved materials cause light to be scattered rather than transmitted in straight lines. An increase in turbidity has been shown to reduce light penetration in lakes and streams and is associated with a decrease in primary production and thus may affect fish through a decrease in some food organisms. Turbidity levels are often expressed as Nephelometric Turbidity Units (NTU) or Jackson Turbidity Units (JTU) which are approximately equal.

Standards applicable to the UK

The Fisheries Directive (CEC 1978) gives a guideline value for EC-designated salmonid fisheries of ≤ 25 mg/l inert SS as an annual average (there is no indication of the sampling frequency). The Directive recognises that floods are liable to cause particularly high concentrations for short periods of time.

No standards in the UK have been adopted for suspended solids nor turbidity for the protection of non-EC-designated salmonids. However, a number of standards that could apply to the protection of UK salmonid fisheries have been proposed, these are outlined below.

EIFAC (Alabaster and Lloyd 1982) proposed tentative criteria for chemically inert solids for the maintenance of fresh water fisheries. They concluded that there was no evidence that a concentration of less than 25 mg/l SS would cause any harmful effects on fisheries. However, they suggested that the spawning grounds of salmon and trout require special considerations, and as such should be kept as free as possible from finely divided solids. In addition, they proposed good or moderate fisheries could be maintained in waters which normally contained 25 - 80 mg/l SS, but those with concentrations greater than 80 mg/l were unlikely to support good fisheries.

Anglian Water Authority (1986) proposed a 95 percentile of less than 80 mg/l or a 99 percentile of less than 150 mg/l SS in salmonid dominated regions. However, they add that normally occurring concentrations are more important than occasional highs, and if SS criteria are to be used to determine a consent condition, a mean water concentration of 25 mg/l should be achieved.

Standards from other countries

Standards for the protection of fisheries that have been set in countries other than the UK include standards for both or either SS and turbidity.

The CCREM (1987) have set a standards for total SS for the protection of aquatic life. This states that the concentration should not exceed 10 mg/l above background when background SS concentrations are equal to or less than 100 mg/l and should not exceed 10% above background concentrations when these are greater than 100 mg/l.

A joint standard for turbidity and SS for the protection of fresh water fish and other aquatic life is recommended by the US EPA (1986). This states that "settleable and SS should not reduce the depth of the compensation point for photosynthetic activity by more than 10% from the seasonally established norm for aquatic life".

In Alaska, standards for turbidity rather than SS are currently used (Lloyd 1987). For waters classified for the growth and propagation of fish, shellfish and wildlife, the standard states that turbidity "shall not exceed 25 NTU above natural conditions level (in streams). For all lake waters, shall not exceed 5 NTU over natural conditions".

Recent studies

A number of studies on the effects of SS and turbidity on salmonids have been reported.

Wilber, 1983 (cited in Bruton 1985) reported that sediment deposition on gravel spawning grounds prevented brown trout *Salmo trutta* from digging redds and caused cutthroat trout *Oncorhynchus clarkii* to abandon their spawning grounds. In addition, sedimentation may cause physical damage to eggs through abrasion or adherence of silt particles to the chorion which may result in their exposure to pathogenic organisms.

Redding *et al* (1987) exposed yearling coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*) to varying concentrations and types of SS. No mortality occurred even at the highest level (4 g/l) but elevated levels of plasma cortisol on exposure to high concentrations of top soil indicated that the exposure may be stressful to the fish. After two days exposure, steelhead showed signs of stress on exposure to all SS at high and low concentrations. The authors suggested that although both coho salmon and steelhead can survive exposure to high concentrations of SS, sublethal stress may effect their performance capacity and thus reduce their capacity to find food or decrease their resistance to disease.

Wilber (1983) (cited in Bruton 1985) reported that high turbidities (1000 - 6000 mg/l) reduced populations of *S. trutta* to one-seventh of their size compared to populations in clear streams. In addition, Wilber (1983) also reported cutthroat trout (*O. clarkii*) stopped feeding and looked for cover after two hours exposure to suspensoid loads of 35 mg/l.

In water with turbidity greater than 30 NTU, avoidance behaviour by rainbow trout and Arctic grayling occurred, (Suchanek *et al*, 1984 cited in Lloyd *et al* 1987). Avoidance was also reported by juvenile coho salmon at 70 NTU and by coho salmon and steelhead in the range 22 - 265 NTU, (Bisson and Bilby 1982, Sigler 1980, Sigler *et al* 1984, cited in Lloyd 1987).

Simmons (1984, cited in Lloyd *et al* 1987) examined the stomachs of Arctic grayling and found that fish in waters of high turbidity contained few or no insects. The authors were unsure if this was due to decrease in feeding due to vision impairment or due to effects on the abundance of macroinvertebrates in such streams.

A reduction in growth rates of *Oncorhynchus mykiss* and *O. kisutch* (Walbaum) in laboratory tests was reported by Sigler *et al* (1984) (cited in Bruton 1985) at turbidity levels as low as 25 NTU.

Lloyd (1987) reviewed the available literature on turbidity in cold water habitats and found it indicated that even low turbidities, 10 - 25 NTU and SS concentrations near 35 ppm can have deleterious effects on fish.

In order to define relationships where turbidity could be used to estimate the concentrations of SS, Lloyd *et al* (1987) used calculations developed for Alaskan streams. The relationship between turbidity and suspended solid concentrations varied among drainages or groups of drainages, but there were definable relationships for Alaskan waters that would enable turbidity to be used as a reasonable indicator of threshold SS levels for a broad range of watersheds in this area.

Lloyd (1987) mentioned current information on the relationships between turbidity and suspended solid concentration to be tentative for a number of reasons: the amount of turbidity produced per unit of suspended solids depends on the sediment, size, shape, etc. and these change with hydrologic or hydraulic conditions; the measurement of turbidity may include some settleable solids depending on the amount of settling occurring at the time of the sample; the relationship between turbidity and SS may change along a downstream gradient from a sediment source; different streams may be able to accommodate different levels of sediment input.

Discussion

The turbidity criteria used in the USA, (US EPA 1986), suffers from several deficiencies. It does not consider the impacts associated with sediment deposition on the bottom, nor the relation to the water column. In addition, it does not address specific levels of SS and places a burden on the authorities to define 'seasonally established norm' for the compensation point. The compensation point was noted by Thurston *et al* (1979) to be of little value in streams where light penetrates to the bottom.

Lloyd (1987) concluded that reasonable turbidity criteria established to protect aquatic habitats from decreased light penetration will also protect them from high concentration of SS. However, separate settleable solids or streambed standards should then be applied to protect aquatic habitats from the impact of heavier sedimentation on benthic substrates. Lloyd *et al* (1987) concluded that the Alaskan water quality standards can be expected to provide a moderate standard for the protection of cold water habitats.

Studies have shown effects of turbidity and SS on salmonids occur as low as 10 NTU and 35 mg/l. The criteria for SS, because they are based on reviews of the data on the effects of fisheries, should be adequate to protect against the physical effects of SS and also against any reduction in light penetration. The strength of turbidity standard lies in its easy measurement as an estimator for both light penetration and SS concentration. However, the relationship between turbidity and SS is difficult to define and a relationship calculated for one catchment will not necessarily hold for the next. Lloyd (1987) and Lloyd *et al* (1987) have shown that adequate turbidity standards should moderately protect fisheries, but additional standards are needed for settleable solids.

It is therefore proposed that the standard set for EC-designated salmonids fisheries and also proposed by EIFAC (Alabaster and Lloyd 1982) not to cause harmful effects on fisheries, i.e. less than or equal to 25 mg SS/l, will adequately protect salmonid fisheries. There are insufficient data to propose a standard for turbidity, further work needs to be carried out before a 'national' standard could be proposed. However, because of the ease of the measurement of turbidity, standards based on local observations may be able to give quick indications of changes and possible harmful effects to fisheries.

2.4 Biological standards

Compliance with fishery objectives could be assessed by monitoring for biological standards (i.e. making a direct assessment of compliance with use). The direct measurement of biological communities would provide a complimentary approach to the measurement of chemical water quality, with the monitoring biological parameters giving a more direct indication of the quality of the aquatic ecosystem.

Gulson (1992) in proposing EQSs to protect general and special ecosystems, discussed a number of techniques that may be suitable for biological assessment of fisheries in rivers and lakes and reported the development of a fish classification system for application throughout the NRA (Mainstone and Wyatt 1991). Since a classification scheme is currently being developed, biological standards for salmonid fisheries will not be proposed here.

2.5 Habitat

Even if water quality is high, fish populations may be reduced or not present at all if the habitat quality has been altered by man's activities.

Habitat alteration, such as cover or shade removal, siltation of gravel due to farming or forestry activities, may affect fish populations while not reducing the chemical quality of the water. In some cases, habitat reconstruction (e.g. laying of artificial gravel beds) may be necessary in order to achieve the fishery objective in waters of otherwise high quality. While it is not in the scope of this report to propose criteria for habitats for salmonid fisheries, it is a necessary parameter to consider when regarding the status of a fishery, with both high habitat and chemical quality needing protecting in order to maintain an adequate fishery.

3. CYPRINID FISHERIES

3.1 Environmental quality objective

This objective applies to fresh water only (rivers and lakes). Parts of the definition (shown below) which are applicable only to rivers are shown in brackets.

EQO	Explanatory notes
Maintain water quality so as to sustain a breeding population of cyprinids appropriate to the physical characteristics (and flow régime) of the water body (where this is not precluded by natural physical barriers).	Generally this will be a sport fishery, not a food source.

As for salmonid fisheries, the EQO for cyprinid fisheries is concerned with all stages of the life cycle, thus stages from eggs to adults need to be considered. Those parameters necessary for maintaining the above objective have been identified from Agg (1989), the EC Fisheries Directive (CEC 1978), Smith *et al* (1991) and Smith and Gulson (1991). Standards set or proposed for the UK and other countries are reviewed, together with available relevant literature, the List II substances reviewed have been limited to the following:

ammonia, arsenic, boron, chromium, copper, inorganic lead, inorganic tin, iron, mothproofing substances, nickel, organotins, pH, sulphide, vanadium and zinc.

The data for all the considered parameters is critically assessed and where possible criteria necessary for maintaining the objective are identified or proposed. Proposed standards, and those applicable to EC-designated cyprinid fisheries are summarised in Tables 3.1 and 3.2.

3.2 Chemical parameters

3.2.1 Dissolved oxygen

As for salmonid fisheries (Section 2.2.3) a number of reviews on the dissolved oxygen requirements for cyprinid fish have been carried out along with proposals for standards for their adequate protection, these are outlined below. In addition, CEC (1978) has set mandatory and guideline values for dissolved oxygen for EC-designated cyprinid fisheries

Standards applicable to the UK

The CEC (1978) has laid down standards for the protection of designated cyprinid fisheries. A mandatory value of 7 mg O₂/l (expressed as 50% of samples) has been set with the proviso that if the concentration falls below 4 mg O₂/l the cause must be investigated and appropriate measures adopted. In addition, guidelines of 8 mg O₂/l (50% of samples) and 5 mg O₂/l (expressed as 100% of samples) are also set.

Table 3.1 Environmental quality standards, proposed and set for the protection of cyprinid fisheries

Parameter	Units	Designated fisheries (A)				Non-designated	
		Guideline value	Statistic	Mandatory value	Statistic	Value	Statistic
Dissolved oxygen	mg O ₂ /l	8 5	50P 100P	7 4	50P INV	5 3	50P 95P
Nitrite	µg N/l	9	95P	-	-	540	95P (1)
BOD	mg O ₂ /l	6	95P	-	-	6	95P (2)
Phosphorous (total)	mg PO ₄ /l	0.4 (3)	-	-	-	-	-
Chlorine	µg HOCl/l	-	-	5	95P	4	MAX
Ammonia (un-ionized)	µg N/l	4	95P	21	95P	21	95P
(total)		160	95P	780	95P	780	95P (G)
pH (4)		-	-	6-9 (5)	95P	6-9	95P
Temperature (difference)	°C	-	-	<1.5	98P	<3.0	MAX
(actual)		-	-	<21.5	98P	28	MAX
(breeding of cold water sp.)		-	-	<10	98P	-	-
Suspended solids (4)	mg/l	25	AA	-	-	25	AA
Arsenic (dissolved)	µg/l	-	-	-	-	50	AA (6)
Boron (total)	mg/l	-	-	-	-	2	AA (6)
Iron (dissolved)	mg/l	-	-	-	-	1	AA (6)
Mothproofing Substances (PCSDs)	µg/l	-	-	-	-	0.05	95P (6)
(sulcofuron)		-	-	-	-	25	95P (6)
(flucofuron)		-	-	-	-	1	95P (6)
(permethrin)		-	-	-	-	0.01	95P (6)
(cyfluthrin)		-	-	-	-	0.001	95P (6)

...../continued

Table 3.1 (continued)

Parameter	Units	Designated fisheries (A)				Non-designated	
		Guideline value	Statistic	Mandatory value	Statistic	Value	Statistic
Organotins TBT	µg/l	-	-	-	-	0.02	MAX (6)
TPT		-	-	-	-	0.02	MAX (6)
Sulphide (as hydrogen sulphide undissociated)	µg/l						
<15°C <5 mg O ₂ /l		-	-	-	-	0.5	AA (7)
<15°C <5 mg O ₂ /l		-	-	-	-	5.0	24A (7)
<15°C >5 mg O ₂ /l		-	-	-	-	1.0	AA (7)
<15°C >5 mg O ₂ /l		-	-	-	-	10.0	24A (7)
>15°C <5 mg O ₂ /l		-	-	-	-	0.25	AA (7)
>15°C <5 mg O ₂ /l		-	-	-	-	2.5	24A (7)
>15°C >5 mg O ₂ /l		-	-	-	-	0.5	AA (7)
>15°C >5 mg O ₂ /l		-	-	-	-	5.0	24A (7)
Inorganic tin (total)	µg/l	-	-	-	-	25	AA (7)
Chromium	}	Standards are related to water hardness, see Table 3.2					
Copper							
Lead							
Nickel							
Vanadium							
Zinc							

...../continued

Table 3.1 (continued)

Notes	(1) Where chloride concentration varies largely from 10 mg/l standard should be varied according to EIFAC criteria.
	(2) Measured as BOD ₅ (ATU) i.e. oxygen demand without nitrification.
	(3) For lakes; value indicative of the need to reduce eutrophication. In the case of lakes of average depth between 18 and 300 m the following formula could be applied: $L < 10 z/Tw (1 + Tw)$ where; L = loading expressed as mg P per square metre lake surface in one year; Z = mean depth of the lake in meters; Tw = theoretical renewal time of the lake in meters.
	(4) Derogation is allowed in the event of exceptional meteorological or geographical conditions.
	(5) Artificial pH variations with respect to the unaffected values shall not exceed ± 0.5 of a pH unit within the limits falling between 6.0 and 9.0 provided that these variations do not increase the harmfulness of other substances present in the water.
	(6) Standards for these List II substances have been adopted as National EQS values (DoE 1989) and these values must be achieved in all waters receiving discharges.
	(7) Standards for these List II substances have been proposed to DoE but not yet adopted as National EQS values. They are suggested as guidelines for the protection of cyprinids to be applied where relevant locally.
	G Guideline.
	I Mandatory.
	P Percentile.
AA	Annual average.
	24A Maximum 24-hour concentration.
	INV Investigation level - when the oxygen concentration falls below this level the cause is to be investigated and the appropriate measures are to be taken.
	MAX Maximum allowable concentration.

References A CEC (1978)

Table 3.2 Hardness related EQSs proposed for the protection of cyprinid fisheries

Substance	Hardness (mg/l as CaCO ₃)						Notes
	<50	50-100	100-150	150-200	200-250	>250	
Chromium	150	175	200	200	250	250	D, AA (1)
Copper	1	6	10	10	10	28	D, AA (1)
Lead	50	125	125	250	250	250	D, AA (1)
Nickel	50	100	150	150	200	200	D, AA (1)
Vanadium	20	20	20	20	60	60	T, AA (1)
Zinc	75	175	250	250	250	500	T, AA (1)

Notes (1) These EQSs have been adopted as National EQS values (DoE 1989) and these values must be achieved in all waters receiving discharges.

AA Annual average.

D Dissolved.

T Total.

No national standards for dissolved oxygen have been adopted in the UK for non-EC-designated cyprinid fisheries, however, a number of standards that are relevant to UK indigenous cyprinids have been proposed.

EIFAC, (Alabaster and Lloyd 1982) have recommended annual 50 and 95 percentile DO values for resident populations of moderately tolerant fresh water species such as roach, of greater than 5 mg/l and 2 mg/l respectively. The values are intended only as guidelines due to seasonal and geographical variations. The authors also state that since early life stages are particularly sensitive the lower DO levels should not occur when these stages are present. In addition, EIFAC proposed tentative minimum steady state DO criteria for normal successful fulfilment of the fish life-cycle under otherwise favourable conditions, these are shown in Table 2.3.

Anglian Water Authority (1986) adopted criteria for the protection of fisheries in its area. For cyprinid fisheries not more than 5% of values (5 percentile) should be below 6 mg/l and 1% of values (1 percentile) below 4 mg/l.

Stiff *et al* (1990) have recently completed a major review of the dissolved oxygen requirements of fresh water fish. They proposed EQSs for the protection of non-EC-designated fisheries, based on the available toxicity and field data. Two EQSs, a median of 5 mg/l DO and a 95 percentile of 2 mg/l, were proposed for adequate protection of coarse fisheries. However, they mentioned that more stringent requirements would have to be met if early life-stages were present. These proposals are the same as the EIFAC criteria.

Milne and Seager (1990) have proposed tentative criteria for dissolved oxygen which take into account the magnitude, duration and return period of pollution events. While such criteria would be valuable for predicting the effects of intermittent pollution events, as mentioned Smith *et al* (1991) there is still a need for validation in the field.

Standards from other countries

A number of standards for dissolved oxygen have been set in countries other than the UK, these are outlined below.

The US EPA have set criteria for ambient oxygen concentrations for the protection of fresh water aquatic life (US EPA 1986) these are presented in Tables 2.4 and 2.5, and have been previously outlined and explained in Section 2.2.3. Criteria set for warm water fishes are applicable to cyprinid fisheries.

Criteria have also been proposed for dissolved oxygen by the National Research Centre for Canada (Davis 1975). These differentiate between fish populations, temperature and levels of risk and are presented in Table 2.6. Three levels of risk are used, A, B, and C. The criteria are based on mean incipient thresholds for documented sublethal effects.

Level A is the level at which few members of the fish community will be likely to exhibit low DO effects (it represents more or less ideal conditions; mean incipient threshold plus one standard deviation).

Level B is the level at which average members of the community may begin to exhibit symptoms of oxygen distress (mean incipient threshold). Some degree of risk to a portion of the population exists at this level if the oxygen minimum period is prolonged beyond a few hours.

Level C is the level at which a large proportion of the pollution may be affected by low DO. It should only be applied to hardy fish populations or fisheries of marginal economic importance and is based on the mean incipient threshold minus one standard deviation.

Discussion

There appear to be no additional data on the effects of dissolved oxygen concentrations on cyprinid fish since Stiff *et al* (1990). Toxicity data cited in Stiff (1990) indicated a wide variation. Lethal levels for adult European 'non-salmonids' were in the range 0.4 - 3 mg/l dissolved oxygen and concentrations affecting survival of embryos in the range 6 mg/l to 2 mg/l. As previously mentioned (Section 2.2.3) Stiff (1990) expanded on the values given by EIFAC (Alabaster and Lloyd 1982) for minimum criteria of DO for maintaining the normal attributes of the life-cycle of fish without significant impairment, these are given in Table 2.7.

Field data from UK NRAs and RPBs cited in Stiff *et al* (1990), indicated coarse fisheries can be sustained at mean and 95 percentile dissolved oxygen concentrations of 7.9 and 4.2 mg/l respectively. This indicates higher concentrations than previously suggested by Alabaster 1973 (cited in Alabaster and Lloyd 1982) where field data indicated a median and 95 percentile dissolved oxygen concentration of 3.7 and 2.1 mg/l respectively would be adequate for the maintenance of a coarse fishery, but higher levels would be necessary if early-life stages were present.

Data presented in the above sections have shown that cyprinid fisheries may be maintained at lower dissolved oxygen concentration than salmonids. Stiff *et al* (1990) reported juvenile and larval survival at 3 mg/l, while Davis (1975) reported populations of cyprinids, while limited, occurred at concentrations as low as 2.5 mg/l. In addition, the US EPA (1986) have proposed an instantaneous criteria of 3 mg/l, that would allow for the maintenance of a cyprinid population. Higher DO concentrations which allow for adequate protection have also been reported. Stiff *et al* (1990) stated growth and long term survival of cyprinids would occur at concentrations greater than 5 mg/l. Davis (1975) stated good populations of cyprinids occurred at 5.5 mg/l while the US EPA (1986) have set a 30 day mean of 5.5 mg/l for the protection of cyprinid populations.

Thus it is proposed that standards that would adequately protect salmonids fisheries would be a 50 percentile of 5 mg/l and a 95 percentile of 3 mg/l. These standards, while the 95 percentile is higher than previously proposed by EIFAC and Stiff *et al* (1990), should, based on the available criteria, reviews and field data, adequately protect all stages of cyprinid populations.

3.2.2 Nitrite

Nitrite (NO_2^-) (an intermediate oxidation state between ammonium and nitrate) is a naturally occurring anion both in fresh and marine waters. Its typical concentration in oxygenated waters being less than 0.005 mg/l (Lewis and Morris 1986).

Eddy *et al* (1983) (cited in Lewis and Morris 1986) stated that nitrite ions are actively taken up by most fresh water fish, and that concentrations in the blood plasma may reach ten times that of the surrounding medium. From the blood plasma, nitrite diffuses into red blood cells where it oxidises the iron in haemoglobin to the 3+ oxidation state, forming methaemoglobin, which lacks the capacity to bind oxygen reversibly. Reductase converts methaemoglobin back to haemoglobin but, this is slow, the normal proportion of haemoglobin being restored in 24 - 48 hours on transferral to water lacking nitrite (Huey *et al* 1980, cited in Lewis and Morris 1986).

Standards applicable to the UK

The EC (CEC 1978) has set a guideline standard for EC-designated cyprinid fisheries of 0.03 mg NO_2 /l as a 95 percentile, this is equivalent to 0.009 mg $\text{NO}_2\text{-N}$ /l.

No environmental quality standards for the protection of non-EC-designated fisheries have been adopted in the UK. However, EIFAC (1984) carried out a review of nitrite toxicity to European fresh water fish and proposed tentative water quality criteria for cyprinids. The criteria vary with respect to chloride concentration since, as previously mentioned (Section 2.2.4), nitrite has been found to be most toxic at low concentrations of chloride. The criteria, proposed as annual averages and 95 percentiles, are given in Table 3.3.

Table 3.3 EIFAC (1984) Tentative nitrite criteria for the protection of coarse fish (mg $\text{NO}_2\text{-N}$ /l)

Chloride (mg/l)	Nitrite (mg $\text{NO}_2\text{-N}$ /l)	
	average	95 percentile
1	0.02	0.06
5	0.10	0.30
10	0.18	0.54
20	0.24	0.72
40	0.30	0.90

Standards from other countries

Both the US EPA (1986) and the CCREM (1987) recommend a concentration at or below 5 mg/l nitrite-nitrogen for the protection of warm water fish.

Recent data

EIFAC (1984) noted that nitrite toxicity is strongly alleviated by chloride ions and that toxicity data indicated that coarse fish, particularly bottom feeding types such as carp and bullheads, are much more resistant to nitrite than salmonids and other related species. Lewis and Morris (1986) found that there was more variation in nitrite toxicity to warm water species of fish (cyprinids) than to cold water fish (salmonids).

Toxicity data indicates that the criteria proposed by EIFAC should be adequate to protect cyprinid species. Solbe *et al* (1985) found the LC50s for roach (*Rutilus rutilus*) and common carp (*Cyprinus carpio*) to be 10 mg NO₂-N/l (20 mg Cl/l) and 15.6 mg/l (19.3 ± 0.3 mg Cl/l) respectively.

Jones and Mallet (1985) found a 14 day LC50 of 2.0 mg NO₂-N/l for the perch *Perca fluviatilis* (259 - 266 mg CaCO₃/l, 14 mg Cl/l) in high DO, while Mallet (1986) reported a 14 day LC50 for *P. fluviatilis* of 2.1 mg NO₂-N/l (260 - 270 mg CaCO₃/l, 12.8 mg Cl/l) in low DO. Indicating that at least for this species, dissolved oxygen concentration appears not to affect the toxicity of nitrite.

Hansan and Macintosh (1986) showed the protective effect of chloride concentration to carp, *Cyprinus carpio*, fry. The seven day LC50 of nitrite increased (i.e. less toxic) as the chloride concentration increased. At 28°C (41 - 49 CaCO₃ mg/l) the seven day LC50 ranged from 2.2 to 42.9 mg NO₂-N/l as the chloride concentration increased from 1.0 to 45 mg Cl/l.

Field data cited in EIFAC (1984) showed that good coarse fisheries occurred at mean nitrite concentrations of below 0.1 mg NO₂-N/l (corresponding to a 95 percentile concentration of 0.3 mg/l). While annual mean chloride concentrations in UK rivers have been reported to be in the range 3 - 100 mg/l (Gardiner and Smith 1990), with typical values being 10 - 30 mg/l except in the Severn Trent and Thames regions where concentrations were higher.

Discussion

The US EPA (1985) and CCREM (1987) based their standards on a review of the toxicity data and cited a study by McCoy (1972) on which they based their recommendation for the protection of warm water fish. McCoy (1972) found that perch (*Perca carpio*) was among the most sensitive cyprinids, surviving less than three hours in 5 mg NO₂-N/l. Data shown here indicate that their recommendation for the protection of warm water fish may not be strict enough at least for UK indigenous cyprinids. In addition, the EC (CEC 1978) standard, which does not consider chloride concentrations, is seven times more strict than that proposed by EIFAC, and the scientific justification behind this cannot be ascertained.

Toxicity data for indigenous cyprinids indicate that the criteria proposed by EIFAC would be sufficient to protect indigenous cyprinids. In view of the data from Gardiner and Smith (1990) on chloride concentrations in UK rivers being typically 10 - 30 mg Cl/l, it is proposed (as in Smith *et al* 1991), that the EIFAC standard of 0.54 mg NO₂-N/l as a 95 percentile at 10 mg Cl/l will adequately protect cyprinid fisheries. If chloride concentrations vary widely from 10 mg Cl/l the standards should vary according to the EIFAC (1984) criteria.

3.2.3 Biochemical oxygen demand

Standards applicable to the UK

The Fresh Water Fish Directive (CEC 1978) has set a guideline value of 6 mg O₂/l for the protection of designated cyprinid fisheries.

While no BOD standards have been set for non-designated cyprinid fisheries, Smith *et al* (1991) recently reviewed and discussed standards that have been proposed and would be relevant for the protection of cyprinid fisheries. These are given in Table 3.4. No additional data since Smith *et al* (1991) are available.

Table 3.4 BOD standards reviewed by Smith *et al* (1991)

Relevant body	Proposed standard for the protection of cyprinids
Anglian Water Authority (1986)	6 mg O ₂ /l (95 percentile)
Baudo <i>et al</i> (1978)	8 mg O ₂ /l
Chiaudani and Premazzi (1988)	10 mg O ₂ /l (mandatory) 7 mg O ₂ /l (guideline)

The standard of 6 mg O₂/l proposed by AWA (1986) is slightly less stringent than that proposed by the EC (CEC 1978) as it is defined as BOD₅ (ATU) that is oxygen demand without nitrification. However, the relevance of the other proposed standards (Baudo *et al* 1978 and Chiaudani and Premazzi 1988) to cyprinid fisheries is difficult to assess as neither stated statistical requirements or gave any derivation.

Since AWA (1986) also stated that some good quality cyprinid fisheries occurred at 95 percentile BOD values of 10 mg O₂/l and due to lack of additional data, it is proposed that the AWA (1986) criteria of 6 mg O₂/l BOD₅ (ATU) be used as a guideline for the protection of cyprinid fisheries. This should be covered with a proviso allowing higher values where it can be shown that DO is not reduced below the relevant criteria.

3.2.4 Phosphorus

As previously stated for salmonid fisheries (Section 2.2.4), phosphate itself is not directly toxic to fish, rather effects are due reduced oxygen concentration that may occur due to algal blooms. It is concluded, as previously stated in Smith *et al* (1991) and as for salmonid fisheries (Section 2.2.4), since eutrophication is likely to have detrimental effects on other water uses before fisheries, a standard for the protection of cyprinid fisheries is not necessary.

For EC-designated fisheries a standard for phosphate concentrations for shallow lakes which 'may be regarded as indicative in order to reduce eutrophication', has been set for EC-designated fisheries in the Fisheries Directive (CEC 1978). In addition, a formula for the calculation of maximum acceptable loading for deeper lakes is given.

3.2.5 Chlorine

In water chlorine undergoes a pH-dependant reaction to produce HOCl which together with Cl_2 , comprises 'free chlorine'. Additional reactions with amines and ammonia result in the formation of chloramines or 'combined chlorine'. Thus the 'total (residual, available) chlorine' comprises of 'free' and 'combined' chlorine. EIFAC (Alabaster and Lloyd 1982) considered that the toxic chemical species in solution in chlorine poisoning is hypochlorous acid (HOCl) and thus the toxicity of any solution containing chlorine will depend on the pH of the solution as this will markedly influence the degree of dissociation.

Standards applicable to the UK

The CEC (1978) Fish Directive gives a mandatory 95 percentile concentration of $<5 \mu\text{g/l}$ total residual chlorine (as HOCl) at pH 6 for EC-designated cyprinid fisheries, higher concentrations of total chlorine can be accepted if the pH is higher. Sampling frequency is specified as monthly. However, it should be noted, that the Directive expresses the standard as 'total residual chlorine', but this is contradicted by stating that it should be expressed as HOCl. It is probable that the standard refers to HOCl.

Standards that may be applicable for UK indigenous fish were proposed by EIFAC (Alabaster and Lloyd 1982). These have been outlined in Section 2.2.5. The proposed standard is $4 \mu\text{g/l}$ HOCl, the corresponding concentrations of total chlorine are given in Table 2.9.

Standards from other countries

Standards for total chlorine for the protection of aquatic life have been proposed by the CCREM (1987) and the US EPA (1986).

The CCREM (1987), recommend that for the protection of fresh water aquatic life the total amount of residual chlorine should not exceed a concentration of $2 \mu\text{g/l}$. While the US EPA (1986) concluded that for the protection of fresh water organisms the four day average concentration of total residual chlorine should not exceed $11 \mu\text{g/l}$ more than once every three years on average and the one hour average concentration should not exceed $19 \mu\text{g/l}$ more than once every three years.

Discussion

No new data on the toxicity of chlorine to cyprinid fish that have not been included in one of the reviews mentioned above appear to be available. Section 2.2.5 discusses these reviews and the criteria that have been proposed.

Criteria proposed for chlorine that apply to cyprinid fisheries are no different to those for salmonids and these have been discussed in Section 2.2.5. The criteria suggested by the CCREM (1987) and the US EPA (1986) appear too stringent and too lax respectively for the protection of UK indigenous fish. It is therefore suggested that, as suggested for non-EC salmonid fisheries, the criteria of a maximum of $4 \mu\text{g HOCl/l}$, as proposed by EIFAC (Alabaster and Lloyd 1982), will be sufficient to protect non-EC-designated cyprinid fisheries.

3.2.6 Ammonia

Standards applicable to the UK

The CEC (1978) in the Fish Directive has set guideline and mandatory values for un-ionized and total ammonia for the protection of designated cyprinid fisheries. As with those set for the protection of designated salmonid fisheries (Section 2.2.6), the standards are expressed as 95 percentiles, samples are to be taken with a minimum frequency of once per month, and the Directive allows that values for un-ionized ammonia may be exceeded in the form of minor peaks in the daytime. A guideline value of 0.005 mg NH_3/l (equivalent to 0.004 mg $\text{NH}_3\text{-N}/\text{l}$) and a mandatory level of 0.025 mg NH_3/l (equivalent to 0.021 mg $\text{NH}_3\text{-N}/\text{l}$) are set for un-ionized ammonia, these are the same as set for designated salmonid fisheries (Section 2.2.6).

Total ammonia standards of 0.2 mg NH_4/l (equivalent to 0.16 mg N/l) (guideline) and 1 mg NH_4/l (equivalent to 0.780 mg N/l) (mandatory) are set 'in order to diminish the risk of toxicity due to non-ionized ammonia, of oxygen consumption due to nitrification and of eutrophication'. The Directive states that 'in particular geographical or climatic conditions and in particular in cases of low water temperature and of reduced nitrification or where the competent authority can prove that there are no harmful consequences for the balanced development of the fish population, member states may fix values higher than 1 mg/l'.

EIFAC (Alabaster and Lloyd 1982) reviewed the available data on ammonia toxicity to fresh water fish and concluded that for cyprinids while laboratory data have shown them to be of similar sensitivity to salmonids, field data have shown that a healthy population can exist where the 50 percentile of un-ionized ammonia is 0.025 mg NH_3/l . They state that while there is evidence that cyprinids can live in waters where concentrations of un-ionized ammonia are higher than the proposed standard, the reason, whether it was due to acclimatisation or low levels of free CO_2 , was not known. In addition, they acknowledge the influence of pH and temperature on the concentration of un-ionized ammonia present and give a table that indicates the concentration of ammonia which at specific temperatures and pH will contain a un-ionized ammonia concentration of 0.025 mg NH_3/l .

Seager *et al* (1988) carried out a major review on the toxicity of ammonia to aquatic organisms but concluded that there was insufficient evidence to justify separate standards for salmonid and cyprinid fisheries. They proposed an EQS of 0.018 mg NH_3/l (equivalent to 0.015 mg $\text{NH}_3\text{-N}/\text{l}$) expressed as an annual average for the protection of fresh water fish in non-EC-designated waters.

Standards from other countries

The EPA (1986) has set water quality criteria for the protection of waters lacking fresh water salmonids and other sensitive cold water species. As for the protection of salmonid species (Section 2.2.6) the criteria use formulae which take into account both pH and temperature to calculate values for one hour and four day exposure to un-ionized and total ammonia which may be exceeded, on average, once every three years. The criteria are presented as a range of acute and chronic limits corresponding to specific combinations of temperature and pH.

Recent data

Since Seager *et al* (1988) there has been very little data on the toxicity of un-ionized ammonia to indigenous cyprinids, with the exception of one study by Mallet (1990). Mallet (1990) investigated the effects of un-ionized ammonia on three species of indigenous fresh water fish, perch (*Perca fluviatilis*), roach (*Rutilus rutilus*) and carp (*Cyprinus carpio*). The median lethal thresholds or asymptotic LC50s for year 1+ fish were found to be in the range 0.23 - 1.6 mg $\text{NH}_3\text{-N/l}$. In terms of mortality, roach were the most sensitive, (median lethal threshold concentrations (MLTC) of 0.23 - 0.35 mg $\text{NH}_3\text{-N/l}$), with perch being of similar sensitivity, (MLTC of 0.46 mg $\text{NH}_3\text{-N/l}$), carp were found to be more resistant (MLTC of 1.2 - 1.6 mg $\text{NH}_3\text{-N/l}$).

Investigating the survival and growth of early life stage of carp and roach, Mallet (1990) found no effects on unhatched eggs at 0.66 and 0.32 mg $\text{NH}_3\text{-N/l}$ respectively. In addition, no difference in sensitivity was seen between starting exposure at six hours post-fertilisation compared with starting on newly hatched fry. Carp early life stages were more sensitive than yearlings (an asymptotic LC50 of 0.5 mg $\text{NH}_3\text{-N/l}$ compared with 1.2 - 1.6 mg $\text{NH}_3\text{-N/l}$ for yearling). For roach this was not the case, early life stages were not more sensitive than older stages (LC50 for larvae, 0.3 - 0.5 mg $\text{NH}_3\text{-N/l}$, for yearling, 0.23 - 0.35 mg $\text{NH}_3\text{-N/l}$). A concentration of 0.32 mg $\text{NH}_3\text{-N/l}$ had no effect on survival over a 130 day exposure period but growth was significantly reduced at this concentration.

Mallet (1990) concluded from reviewed data and his own experiments that while salmonids are generally considered to be more sensitive to pollutants than cyprinids, at least for older life-stages, this does not appear to be the case for ammonia.

Hermenutz *et al* (1987), reviewed and cited a number of studies for un-ionized ammonia toxicity for non-indigenous cyprinid species. Sub-lethal effects were observed in the range 0.011 - 0.297 mg $\text{NH}_3\text{/l}$; fathead minnows (range 0.091 - 0.297 mg $\text{NH}_3\text{/l}$); channel catfish (range 0.011 - 0.153 mg $\text{NH}_3\text{/l}$); walleye (range 0.099 - 0.268 mg $\text{NH}_3\text{/l}$) and white sucker (30 day study causing swim up delay and length reduction, 0.058 - 0.068 mg $\text{NH}_3\text{/l}$). The 96 hour LC50s for white sucker and walleye were reported in the range 0.36 - 2.22 mg $\text{NH}_3\text{/l}$. While a 30 day early life stage test on channel catfish indicated significant reduction in weight at 0.20 mg $\text{NH}_3\text{/l}$ and 96 hour LC50s for white sucker and walleye were reported in the range 0.36 - 2.22 mg $\text{NH}_3\text{/l}$.

Field data, (Seager *et al* 1988) showed satisfactory cyprinid fisheries exist at a number of sites where un-ionized ammonia exceeded the EIFAC criteria. Severn Trent Water, found that at 42 given sites at which the 95 percentile concentration of un-ionized ammonia was greater than 0.021 mg $\text{NH}_3\text{-N/l}$, two supported good cyprinid fisheries, and 23 had average or moderate fisheries. In addition, a good mixed fishery was reported by Northumbrian Water where mean and maximum un-ionized ammonia values were 0.02 and 0.05 $\text{NH}_3\text{-N/l}$.

Discussion

EIFAC (Alabaster and Lloyd 1982) concluded that prolonged exposure to ammonia was as equally toxic to certain cyprinid fisheries as to salmonids. However, Seager *et al* (1988) stated that there was some evidence of cyprinids being more resistant to ammonia than

salmonids over short periods, but there was a need for further work on chronic exposure to early life stages of cyprinids before they could justify the setting of separate standards for salmonid and cyprinid fisheries. Their proposed EQS for fish was based on early life stage data of rainbow trout from Solbe and Shurben (1989).

However, Mallet (1990) has since found early life stage LC50s for two indigenous cyprinids to be in the range 0.3 - 0.5 mg $\text{NH}_3\text{-N/l}$ and for three indigenous cyprinid species the yearling LC50 are in the range 0.23 - 1.6 mg $\text{NH}_3\text{-N/l}$. He stated that the toxicity of ammonia to early life stages of carp and roach was similar to other studies, with the exception of Solbe and Shurben 1989.

Thus while Mallet (1990) has indicated that early life stages of indigenous cyprinids do not appear to be as sensitive as older life stages, as mentioned in Section 2.2.6, pH and temperature have also been found to affect the toxicity of ammonia, although the exact relationship is still unclear. Because of the uncertainty of this relationship, the ammonia standards proposed in Section 2.2.6 for salmonid fisheries were ultimately based on field observations. Field observations of cyprinids also indicate that a 95 percentile of 0.021 mg $\text{NH}_3\text{-N/l}$ would allow for good cyprinid fisheries and thus it is proposed that this standard, 0.021 mg $\text{NH}_3\text{-N/l}$ (0.025 mg $\text{NH}_3\text{-N/l}$) (as a 95 percentile) the same as the EC mandatory standards, would be adequate for the protection of cyprinid fisheries. Higher levels should be allowed if found not to be detrimental.

As mentioned in Section 2.2.6, Smith and Gulson (1991) discussed the need for standards for total ammonia to provide a firm basis for the calculation of discharge consents. They concluded that due to differences in ambient pH and temperature characteristics between rivers, further research was needed before a meaningful standard for total ammonia, which takes into account these factors in the receiving water, could be set. They therefore suggested that the mandatory standard for total ammonia for EC-designated cyprinid fisheries be adopted as an interim standard for non-EC-designated cyprinid fisheries, although they noted that this may be too stringent in some cyprinid waters. It is therefore proposed that the EC mandatory total ammonia standard of 1 mg $\text{NH}_4\text{-N/l}$ (equivalent to 0.780 mg N/l) be used as a guideline standard for the protection of cyprinid fish with the added proviso allowing higher values where it can be shown that there are no deleterious effects on cyprinid fish populations.

3.2.7 Arsenic

Arsenic may be released into water from mining, smelting and refining and is also used as a wood preservative and in the manufacture of glass, alloys or medicines.

Standards applicable to the UK

Mance *et al* (1984a) proposed an EQS of a dissolved annual average of 50 $\mu\text{g As/l}$. This has since been adopted by the Department of the Environment (DoE 1989) for the protection of aquatic life (e.g. cyprinids). Mance *et al* (1984a) did not suggest separate standards for trivalent and pentavalent arsenic, despite information that the trivalent form may be more toxic to aquatic life, because a meaningful comparison of the two forms was not possible. A review of the EQS is currently being carried out by WRc for the Department of the Environment.

Standards from other countries

The US EPA (1986) recommended standards for the protection of fresh water aquatic organisms in the form of arsenic (III). They state that they should not be affected unacceptably if the four day average concentration of arsenic (III) does not exceed 190 µg/l more than once every three years on average and if the one hour average concentration does not exceed 360 µg/l more than once every three years. Standards for the pentavalent form were not proposed due to lack of data.

The CCREM (1987) recommend a guideline of 0.05 mg/l total arsenic for the protection of aquatic life. The guideline is stated to be well below concentrations known to be toxic to sensitive early life stages.

Discussion

The toxicity data that was the basis of the standards proposed by Mance *et al* (1984a) and the CCREM (1987) has previously been outlined in Section 2.2.7.

Available data on the toxicity of arsenic to indigenous cyprinids since Mance *et al* (1984a) are limited to one study. Mallet (1990) investigated the acute toxicity of arsenite (trivalent form) and arsenate (pentavalent form) to the roach and concluded that arsenite (III) was an order of magnitude more toxic than arsenate (V). For arsenate (V), the 96 hour LC50 was in the range 170 to 200 mg/l while, for arsenite (III) the 96 hour was 11.8 mg/l. However, for arsenite, the threshold value had still not been reached after 96 hours (i.e. the LC50 was still decreasing).

Hence while recent toxicity data from Mallet (1990) indicate that arsenite (III) may be an order of magnitude more toxic to roach than arsenate (V) the LC50 indicate that the EQS of 50 µg/l dissolved annual average of arsenic should be adequate for the protection of cyprinid fish. Thus it is proposed that the current EQS of 50 µg As/l (dissolved annual average) based on the available data will be adequate for the protection of cyprinid fisheries.

3.2.8 Boron

Standards applicable to the UK

An environmental quality standard of 2 mg/l as an annual average for total boron, for the protection of all fresh water fish, irrespective of water hardness, was proposed by Mance *et al* (1988a). This standard has since been adopted (DoE 1989) for the protection of cyprinid fish.

Standards from other countries

No other country appears to have set or proposed standards for boron for the protection of fresh water fish or all aquatic organisms.

Discussion

Data reported in Mance *et al* (1988a) on the toxicity of boron to cyprinid fish were very limited. The only reported test on an indigenous fish was on *Rutilus rutilus* (roach) by Wurtz in 1945.

Since Mance *et al* (1988a) the only available data on toxicity of boron to cyprinids were reported by Butterwick *et al* (1989) who reviewed the safety of boron in aquatic and terrestrial environments including a review of the available data on the toxicity of boron to fish. However, the majority of tests cited were previously used for the derivation of the EQS by Mance *et al* (1988a) and additional tests were on non-indigenous cyprinids.

Butterwick *et al* (1989) stated early life stage non-salmonids appear to be relatively resistant to aqueous boron exposure and cited Avetisyan (1983) who found the addition of 0.4 mg B/l to a pond used for rearing carp increased production by 7.6%, indicating that low concentrations of boron may even be beneficial. Only one acute toxicity test was not cited in Mance *et al* (1988a). Birge and Black (1981) found the 11 day NOEC and LOEC of freshly fertilised large mouth bass (*Micropterus salmoides*) eggs in reconstituted hard water to be 1.39 - 12.17 mg B/l, however this is a very large gap between concentrations.

Environmental data in Mance *et al* (1988a) reported from Thames Water Authority, indicated good coarse fisheries can exist in rivers of intermediate water hardness at boron concentrations of 0.4 - 0.5 mg/l.

Additional data since Mance *et al* (1988a) for cyprinids are limited to one test, this does not give any indication that the EQS of 2 mg/l will not adequately protect cyprinid fisheries. Thus it is proposed that at present, this EQS appears adequate to protect cyprinid fisheries.

3.2.9 Chromium

Standards applicable to the UK

Mance *et al* (1984b) proposed annual average EQSs for dissolved chromium VI, with respect to water hardness, for the protection of cyprinid fisheries (Table 3.5). These have since been adopted (DoE 1989) for the protection of cyprinids, but as annual average dissolved concentrations of chromium III and VI. Mance *et al* (1984b) did not propose separate EQSs for chromium III due to lack of relevant data.

Table 3.5 EQSs for the protection of cyprinid fisheries (DoE 1989)

Total water hardness (mg CaCO ₃ /l)	Annual average dissolved chromium III and VI (µg Cr/l)
Less than 50	150
50 - 100	175
100 - 200	200
Greater than 200	250

The EQSs are currently being reviewed by WRc for the DoE.

EIFAC (1983) reviewed data on the effects of chromium to European fresh water fish and proposed stricter standards for cyprinids than adopted by the DoE (1989). It was proposed that the mean concentrations of chromium should not exceed 100 µg/l and the 95 percentile should not exceed 400 µg/l. They stated that high concentrations should not occur during the breeding season.

Standards from other countries

Standards set by the US EPA (1986) and the CCREM (1987) for the protection of aquatic life have previously been outlined in Section 2.2.9. The CCREM (1987) set a guideline of 0.02 mg/l total chromium not to be exceeded. In contrast, the US EPA (1986) set fixed values for Cr VI that should not be exceeded, while values for Cr III are calculated from formulae based on water hardness.

Discussion

Since Mance *et al* (1984b) only two tests on the toxicity of chromium to cyprinids have become available.

Chromium III was found by Wong *et al* (1982) to be of low toxicity to carp (*Cyprinus carpio*) with significant mortality only occurring at concentrations greater than 20 mg/l. Jenkins and Willis (1982) using flow through tests in hard water (257 mg CaCO₃/l) over 12 weeks, found the NOEC for growth to be 0.75 mg Cr/l, the oxidation state of chromium was not reported.

Mance *et al* (1984b) compared environmental concentrations of chromium VI in UK rivers, it was indicated that the EQSs for coarse fish are not approached even by total chromium (particulate, and dissolved Cr VI and Cr III).

Since additional toxicity data for cyprinids since Mance *et al* (1984b) are not available, it is difficult to assess if the proposed EQS for chromium VI is other than adequate for the protection of cyprinids. While new toxicity data for chromium III are available it is insufficient to propose a standard. It is therefore proposed that the current EQSs adopted by the DoE (1989) for both chromium III and VI should adequately protect cyprinid fisheries.

3.2.10 Copper

Standards applicable to the UK

CEC (1978) have recommended guideline standards for the protection of EC-designated cyprinid fisheries, the guidelines vary with water hardness and are the same as recommended for the protection of EC-designated salmonid fisheries (see Section 2.2.10).

In addition, EQSs for copper for the protection of fresh water fisheries, were proposed by Mance *et al* (1984c) and have since been adopted (DoE 1989) for the protection of cyprinids. The standards are related to water hardness and as for the CEC, are the same as adopted for the protection of salmonids, they have previously been outlined in Section 2.2.10.

The EQSs for copper are currently being reviewed by WRc and thus additional data since Mance *et al* (1984c) have not been presented. Any new proposed standards will undergo a period of consultation with the Department of the Environment before being adopted.

Standards from other countries

Standards for the protection of aquatic life have been proposed by the US EPA (1986) and the CCREM (1987). These have been outlined in Section 2.2.10. Both propose standards based on water hardness. The US EPA (1986) use formulae to calculate four day and one hour standards, and the CCREM (1987), with the exception of the standard for soft waters, use the US EPA hardness formula for the four day standard multiplied by an additional factor because they believed the effect of hardness on chronic toxicity to be inconclusive.

Discussion

Standards for copper proposed for the protection of cyprinid fisheries are no different from those proposed for salmonids which are previously discussed in Section 2.2.10. As no additional data are presented and because a review of the EQSs is currently being undertaken, it is proposed that the EQS adopted is currently adequate for the protection of cyprinid fisheries.

3.2.11 Inorganic lead

Standards applicable to the UK

Environmental quality standards for annual average dissolved concentrations of lead have been proposed by Brown *et al* (1984) for the protection of cyprinids. These have since been adopted by the Department of the Environment (DoE 1989). The standards are related to water hardness and are given in Table 3.6. Brown *et al* (1984) stated that other values may be appropriate to local situations such as the protection of very sensitive species, or when organolead is present as a significant proportion of the dissolved lead. The EQSs are currently under review by WRc for the DoE.

Table 3.6 Environmental quality standards proposed by Brown *et al* (1984) for the protection of cyprinid fisheries

Water hardness (mg/l CaCO ₃)	Annual average dissolved concentration of inorganic lead (µg/l)
<50	50
50 - 150	125
>150	250

Standards from other countries

Standards for the protection of fresh water aquatic life have been set by the US EPA (1986) and the CCREM, (1987). These have been outlined in Section 2.2.11. Both the US EPA (1986) and CCREM (1987) standards are based on calculations involving water hardness and are stricter than the adopted EQSs (DOE 1989).

Discussion

Data on indigenous coarse fish in Brown *et al* (1984) were restricted to one study on pike (*Esox lucius*). However, lead was found to be more toxic to three non-indigenous species and thus the EQS was based on these less tolerant species. There were no data for hard waters, the EQS was calculated by comparison of the ratios for the EQSs for salmonid waters. No additional data on the toxicity of lead to cyprinid species since Brown *et al* (1984) appear to be available.

Field data in Brown *et al* (1984) indicated that concentrations of lead in UK surface waters were generally lower than the EQS proposed for cyprinids.

The US EPA (1986) criteria, as previously discussed in Section 2.2.11 are difficult to compare with the EQSs as they are for total lead and the database on which the hardness formulae are based is not given, thus their relevance to UK indigenous cyprinids is hard to compare. Because no additional data since the derivation of the EQSs are available, it can only be proposed that the current adopted EQS for cyprinid fish is adequate. However, the data on which these standards were proposed are very limited, and considerably more data for acute and chronic exposure, for all life stages and in different water hardnesses are needed.

3.2.12 Inorganic tin

Standards applicable to the UK

A tentative EQS for inorganic tin, as an annual average concentration of 25 µg total Sn/l has been proposed by Mance *et al* (1988b) for the protection of fresh water life, and thus this standard should protect cyprinid fisheries. The EQS has yet to be adopted by the Department of the Environment.

Standards from other countries

There do not appear to be any standards for inorganic tin proposed by other countries for the protection of cyprinid fisheries.

Discussion

The tentative EQS proposed by Mance *et al* (1988b) for the protection of all fresh water aquatic life was based on a single study of the invertebrate *Gammarus pulex*, rather than on fish toxicity data. Mance *et al* (1988b) found available cyprinid toxicity data extremely limited, and the reported studies were only on non-indigenous fish. Since this review, there do not appear to be any additional data on the toxicity of inorganic tin to cyprinid fish.

Data on environmental concentrations are also very limited. Mance *et al* (1988b) only reported data on concentrations of inorganic tin from North American fresh waters. However, these showed that concentrations of inorganic tin appeared to be limited to nanograms/l concentrations.

Because of the limited data, Mance *et al* (1988b) were unable to recommend separate EQSs for cyprinid fisheries. Since there appears to be no additional data on the effects of inorganic tin of cyprinid fish, it is still not possible to recommend a separate environmental quality standard for their protection. The previous standard recommended by Mance *et al* (1988b) for the protection of all fresh water life appears adequate considering the paucity of the data.

3.2.13 Iron

Standards applicable to the UK

An EQS for total iron of 2 mg Fe/l for the protection of fresh water fish was proposed by Mance and Campbell (1988). After consultation with the Department of the Environment, an annual average standard of 1 mg/l dissolved iron has since been adopted (DoE 1989) for the protection of fresh water life.

Standards from other countries

As previously mentioned in Section 2.2.13 both the US EPA (1986) and CCREM (1987) have recommended levels of total iron for the protection of fresh water aquatic life of 1 mg Fe/l and 0.3 mg Fe/l respectively.

Discussion

Mance and Campbell (1988) concluded that the available toxicity data did not show a difference in the toxicity of iron to salmonid or cyprinid fish and thus separate standards were not proposed. In addition, no variance in toxicity with respect to different water hardnesses was seen, however, below pH 7, the toxicity of iron increases and they stated that controlling authorities may need to reduce the EQS.

No additional data since the review by Mance and Campbell (1988) on the toxicity of iron to indigenous cyprinids appear to be available. The relevant data have been discussed in

Section 2.2.13. The same conclusions for cyprinids as for salmonids can be drawn, i.e. that the EQS proposed by Mance and Campbell (1988) and since adopted by the DoE (1989) will be adequate for the protection of cyprinid fisheries.

3.2.14 Mothproofing substances

Standards applicable to the UK

EQSs were proposed by Zabel *et al* (1988a) for four chemicals used in mothproofing, chlorphenylid, sulcofuron, flucofuron and permethrin (see Section 2.2.14) for the protection of fresh water fish. EQSs could not be proposed for two other chemicals, cyfluthrin and HHP. These standards have since been adopted by the DoE (1989) for the protection of fresh water life. Further information is currently limited and the reader is referred to the review by Zabel *et al* (1988a) for information on the derivation of these standards.

3.2.15 Nickel

Standards applicable to the UK

EQSs for nickel, with respect to water hardness, for the protection of all fresh water fish were proposed by Mance and Yates (1984a), which have now been adopted (DoE 1989) for the protection of cyprinid fish. These are given in Section 2.2.15.

The EQSs are currently under review by WRc for the DoE.

Standards from other countries

The US EPA (1986) have proposed a 24 hour average guideline for the protection of fresh water aquatic life, based on a formula concerned with water hardness and a formula, and a maximum value not to be exceeded, also based on water hardness. The formulae, along with calculated examples are given in Section 2.2.15.

The CCREM (1987) have also proposed standards, with respect to water hardness, for the protection of aquatic life. These are given in Section 2.2.15. The standards are based on a guideline recommended by Taylor *et al* (1979) and the US EPA (1986) 24 hour formula.

Discussion

As previously mentioned (Section 2.2.15) the adopted EQSs for soft and hard water were based on laboratory data from non-indigenous fish, while that for medium water hardness was calculated by interpolation. Separate standards were not proposed for cyprinids due to lack of data, and the authors concluded from the limited data that cyprinid and salmonid species appeared to be of similar sensitivity. As previously discussed for salmonid fisheries, the integrity of the formulae on which the US EPA (1986) and the CCREM (1987) standards are based have been called into question.

Additional data on the toxicity of nickel to cyprinids are limited to one test. Muramoto (1983), found 48 hour exposure to 20 mg Ni/l caused 50% mortality of *Cyprinus carpio*. However, no water hardness was given. Since this additional data gave no information on

water hardness, its relevance to the EQS cannot be assessed. It is therefore proposed, that the current EQSs (DoE 1989) are adequate for the protection of cyprinid fisheries.

3.2.16 Organotins

Standards applicable to the UK

EQSs of 20 ng/l (total), as 95 percentiles, for tributyltin and triphenyltin for the protection of fresh water fish have recently been proposed by Zabel *et al* (1988b). Since their proposal, the DoE (1989) has adopted EQSs of 20 ng/l (total) as a maximum allowable concentration for TBT and TPT for the protection of fresh water aquatic life which includes cyprinids.

Discussion

As previously mentioned in Section 2.2.16, the EQSs proposed by Zabel *et al* (1988b) for TBT and TPT were very tentative, due to the lack of chronic and field data. For tricyclohexyltin data were limited to one acute toxicity test and hence no EQS for the protection of fresh water fish could be proposed. As crustaceans and molluscs tend to be the most sensitive species to organotins, the EQSs derived for the protection of fresh water life should be adequate for the protection of cyprinid fish.

3.2.17 pH

The literature on the effects of pH to fresh water fish has previously been reviewed in Section 2.2.17. None of the reviews have found any differences in the effects of pH on salmonid or cyprinid fisheries and thus the literature and standards outlined in Section 2.2.17 are of equal relevance to cyprinid fisheries.

pH standards that specifically relate to cyprinid fisheries, but are not different to that proposed for salmonid fisheries are outlined below.

Standards applicable to the UK

The Fisheries Directive (CEC 1978) has set a mandatory level of pH 6 - 9 as a 95 percentile for the protection of EC-designated cyprinid fisheries (the same as for EC-designated salmonid fisheries). As with salmonid fisheries, the sampling should be monthly with a reduction in the frequency allowed where the water is known to be well within the limits specified, pollution is absent, or where there is no risk of deterioration in the water quality. The range is based on the assumption that concentrations of other potentially harmful substances are low. The Directive also sets a condition that artificial variations of pH with respect to background levels should not exceed ± 0.5 pH units within the mandatory levels and changes should not increase the harmfulness of other substances in the water. Derogation is allowed in the event of exceptional weather, geographical conditions or natural enrichment.

Wolff *et al* (1988) have recently carried out a major review on the effects of pH on fisheries. They proposed an EQS for the protection of fresh water fish of pH 6.0 - 9.0 also as a 95 percentile (monthly sampling). This is the same as set by the EC (CEC 1978) and has since been adopted by the DoE (1989) for the protection of fresh water life.

Discussion

Since the effect of pH on cyprinids has not been found to be different to that on salmonids, it can be concluded that the present EQS for the protection of cyprinid fisheries, pH 6 - 9 (as a 95 percentile on an annual basis) (DoE 1989) is adequate for the protection of resident cyprinid fisheries.

3.2.18 Sulphide

Standards applicable to the UK

As mentioned in Section 2.2.18, EQSs for sulphide expressed as undissociated hydrogen sulphide with respect to temperature and oxygen concentration have been proposed by Mance *et al* (1988c) for the protection of fresh water fish. However, these have yet to be adopted by the DoE, and a review of the toxicity data and expression of the EQSs is currently being carried out by WRc for the DoE.

Discussion

The EQS for sulphide proposed by Mance *et al* (1988c) has been outlined in Section 2.2.18. Since the EQS is proposed for the protection of fresh water fish, it should be adequate for the protection of cyprinid fisheries. It is therefore proposed that the sulphide standards proposed by Mance *et al* (1988c) be applied for the protection of cyprinid fisheries, until further information, expected in the review currently being carried out is available.

3.2.19 Vanadium

Standards applicable to the UK

Environmental quality standards for total vanadium, with respect to water hardness, for the protection of fresh water fish were proposed by Mance *et al* (1988d) and have since been adopted by the Department of the Environment for the protection of cyprinid fish. They are given in Table 3.7.

Table 3.7 Environmental quality standards for total vanadium for the protection of fresh water fish (Mance *et al* 1988d, DoE 1989)

Water hardness (mg/l CaCO ₃)	Annual average concentration (µg V/l)
< 200	20
> 200	60

Standards from other countries

There appear to be no standards set or proposed by other countries for vanadium for the protection of fresh water fish, or more specifically for the protection of cyprinid fisheries.

Discussion

Data in Mance *et al* (1988d) on vanadium toxicity to cyprinids are limited, the majority being for non-indigenous fish. A review of the data since this review has not indicated any additional data on its toxicity to indigenous cyprinids.

Field data reported by Mance *et al* (1988d) from Northumbrian WA region indicated that at the highest concentrations reported in rivers, 20 and 40 µg/l (water hardness not stated), both supported good sport fisheries and it is probable that such concentrations would also support a cyprinid population. From the available data it is only possible to conclude that if the EQSs are met cyprinid fisheries should be adequately protected.

3.2.20 Zinc

Standards applicable to the UK

Standards for EC-designated cyprinid fisheries have been set by the (CEC 1978) these are given in Table 3.8. The standards are mandatory and expressed as 95 percentiles of total zinc with a minimum sampling frequency of one sample per month. The Directive states a derogation, allowing the standards to be exceeded if caused by natural enrichment e.g. discharges from old mine workings.

Table 3.8 Mandatory values for EC-designated cyprinid fisheries (CEC 1978)

Water hardness (mg/l CaCO ₃)	Total zinc (µg/l) 95 percentile
10	300
50	700
100	1000
500	2000

In addition, environmental quality standards for total zinc for the protection of non-EC-designated cyprinid fisheries were proposed by Mance and Yates (1984b) and have since been adopted by the DoE (1989) for the protection of cyprinid fish. Most other EQSs for metals have been set as dissolved concentration rather than as here, total concentration. The EQSs are annual averages varying with respect to water hardness. They are the same as originally proposed by EIFAC (Alabaster and Lloyd 1982) and are given in Table 3.9. The EQSs are currently under review by WRc for the Department of the Environment.

Table 3.9 EQSs for the protection of cyprinid fish (Mance and Yates 1984b)

Water hardness (mg/l CaCO ₃)	Total zinc (µg/l) (annual average)
10	75
50	175
100	250
500	500

Standards from other countries

The US EPA (1987) have set one hour and four day average zinc concentrations, that should not be exceeded more than once every three years on average, for the protection of aquatic life. These have been outlined, along with example calculations, in Section 2.2.20.

The CCREM (1987) give what they state is a tentative guideline of 0.03 mg/l total zinc that should not be exceeded. The standard is stated to coincide with the measured 'no effect' concentration for rainbow trout and fathead minnow (0.036 and 0.03 mg/l respectively). It does not vary with water hardness since there were believed to be insufficient data to indicate chronic toxicity decreases as water hardness increases.

Discussion

Data since Mance and Yates (1984b) on the toxicity of zinc are limited to one test and this is for non-indigenous species. Carlson *et al* (1986) reported 96 hour LC50 of 0.12 mg/l and 0.43 mg/l for young striped bass (*Morone saxatilis*) in hard and soft water. In addition, they found that the lowest adverse effect concentration for fathead minnow (*Pimephales promelas*) in soft water was a 96 hour LC50 of 0.4 mg/l.

As previously mentioned (Section 2.2.20) the toxicity of zinc appears to increase in soft waters while toxicity data for cyprinids since Mance and Yates (1984b) are limited, those available indicate that adequate protection would be afforded by the current EQSs. It is therefore proposed that these standards should adequately protect cyprinid fisheries.

3.3 Physical parameters

3.3.1 Temperature

The tolerance of fresh water fish to high temperatures depends on a number of factors including the species, stage of development, season, acclimation temperature, concentrations of dissolved oxygen and the presence of pollutants. Standards for temperature for the protection of fisheries are necessary since changes in the temperature régime may produce changes in the population composition, and fish behaviour.

Standards applicable to the UK

With the exception of standards for EC-designated cyprinid fisheries, no standards for temperature have been adopted in the UK for cyprinid fisheries.

The Fisheries Directive (CEC 1978) gives three mandatory temperature standards, (98 percentiles) concerned with thermal discharges in designated cyprinid fisheries. The temperature measured downstream of a point of discharge at the edge of a mixing zone must not exceed the unaffected water temperature by more than 3°C, derogations may be allowed for conditions where the competent authority can prove that there are no harmful consequences for the balanced development of the fish population. In addition, thermal discharges must not cause the temperature downstream of the point of thermal discharge (at the edge of the mixing zone) to exceed 28°C or 10°C during breeding periods in water containing species that need cold water for reproduction.

Proposed standards for temperature that may be suitable for cyprinid fisheries have been outlined below.

EIFAC (Alabaster and Lloyd 1982) tentatively recommended water temperature guidelines for cyprinids, based on a review of the literature of European fish, these are divided into three seasons:

1. Autumn and winter: for pike, percids and cyprinids a temperature increase of 5 - 6°C above the norm may accelerate maturation of gonads and cause spawning much earlier in the spring. An increase in the temperature of 2°C above the norm in autumn and winter at the time of reproduction of burbot (*Lota lota*) would be damaging to the reproduction of this species although the majority of other species would be unaffected.
2. Spring: for most spring-spawning fish the optimum range of temperature for spawning and embryonic development is no greater than 8°C.
3. Summer: they recommended that for cyprinids it would be permissible to increase to 6°C above ambient, with an upper limit of 30°C, however, they noted that at 28°C the growth of several cyprinids is inhibited.

Anglian Water Authority (1986) have proposed water quality criteria for temperature in cyprinid dominated rivers. These are a 95 percentile of 25°C and a 99 percentile of 28°C.

Standards from other countries

The US EPA (1986) have set temperature criteria for the protection of all aquatic life. As previously mentioned (Section 2.3.1), the criteria are based on two limits, a maximum temperature for short exposure, which is time dependent and based on a species specific equation, and a limit based on the weekly average temperature, that may be based on one of four criteria (see Section 2.3.1).

Calculated examples for non-salmonids, of short-term temperature maxima for survival for juveniles and adults during the summer, maximum average weekly temperatures (MAWT) for growth (b) along with reported optimal temperatures for spawning and upper temperature

for successful incubation and hatching (ERL-Duluth 1976, reported in US EPA 1986) are given in Table 3.10.

Table 3.10 Calculated examples of US EPA (1986) temperature criteria

Species	Short term maxima °C	MAWT (Growth)(b) °C	Reported optimum temperature °C spawning	Reported optimum temperature °C incubation /hatching
Carp	nr	nr	21	33
Channel catfish	35	32	27	29
Northern pike	30	28	11	19
Yellow perch	nr	29	12	20

Recent data

Recent available literature for the effect of temperature on cyprinids have been reviewed. However, only limited data were found to be available, these are outlined below.

Early-life stages

Herzig and Winkler (1986) investigated the influence of temperature on the embryonic development of three European cyprinid fish, *Abramis brama*, *Chalcalburnus chalcoides mento* and *Vimba vimba*. Hatching success results indicating lower and upper lethal temperature of around 9.0 - 9.5 and 23 - 24°C for *C. chalcoides*, 10 - 11 and 23.5 - 24°C for *V. vimba* while for *A. brama* hatching success was 90 - 99% at 10.5 - 16.2°C. The author indicated that the range of tolerated temperature was in good agreement with temperatures at which the various cyprinid species spawned.

Adults

Kilgour *et al* (1985) cited observed upper incipient lethal temperatures (UILT). For *Ictalurus nebulosus* (catfish species) acclimated to 20 and 25°C respectively, UILTs were 33 and 34°C respectively, while for *Perca flavescens* UILTs of 30 and 32°C were cited for fish acclimated to 25°C.

Discussion

Compared to data on the effects of temperature on salmonids, those for cyprinids are limited, however, the data and water quality standards proposed by various authorities show that cyprinids can tolerate higher temperatures than salmonids. As with salmonids, it is impossible to set standards that will protect each life stage, rather it is more practical to set standards above which there may be an adverse effect on the population. Therefore based on

the literature and previously mentioned standards, a maximum temperature of 28°C is proposed for the protection of cyprinid fisheries. This maximum should only apply in waters affected by thermal discharges: natural temperature may exceed these values.

As for salmonid fisheries, in addition to the proposed standard for maximum temperature, a standard for maximum temperature change due to thermal discharges is also proposed. The standard is for a temperature difference (up or down) from unaffected. It is proposed that the EC Fish Directive standard for temperature difference be applied to non-designated waters also. That is, the temperature difference at the edge of the mixing zone, compared to unaffected water, should not exceed 3°C. This could be expressed as a maximum allowable difference which would be a more practicable expression than the 98 percentile in the Directive.

3.3.2 Suspended solids and turbidity

Increased suspended solids (SS) may cause a number of effects on fish. These may be indirect such as blanketing of the stream bed, which may effect the availability of food; blocking of gravel which may hinder spawning or cause a reduction in the flow of oxygenated water which may in turn affect the survival and development of eggs or parr, or direct such as siltation of gills.

Certain factors may increase the concentration of SS these include, afforestation, during the early stages of the forestry cycle and felling, and gravel extraction, which may cause the release of fine sediments.

Turbidity is an optical property of water whereby suspended and dissolved materials cause light to be scattered rather than transmitted in straight lines. An increase in turbidity has been shown to reduce light penetration in lakes and streams and is associated with decrease in primary production and thus may affect fish through a decrease in some food organisms. Turbidity levels are often expressed as Nephelometric Turbidity Units (NTU) or Jackson Turbidity Units (JTU) which are approximately equal.

Standards applicable to the UK

The Fish Directive (CEC 1978) has set a guideline value for inert SS as ≤ 25 mg/l as an average concentration for designated cyprinid fisheries (no indication on the sampling frequency is given). The Directive recognises that floods are liable to cause particularly high concentrations of SS for short periods of time.

As for salmonid fisheries (Section 2.3.2) no national standards for SS or turbidity for the protection of cyprinid fisheries have been adopted in the UK. However, two standards for SS, that may be relevant to UK cyprinid fisheries, have been proposed.

As previously mentioned in Section 2.3.2, EIFAC (Alabaster and Lloyd 1982) concluded that there was no evidence that concentrations less than 25 mg SS/l would be harmful to fresh water fisheries. In addition, Anglian Water Authority (1986) have proposed two SS standards for cyprinid dominated regions, a 95 percentile of less than 100 mg/l or a 99 percentile of less than 200 mg/l. However, they add that normally occurring concentrations

are more important than occasional highs, and if suspended solid criteria are used to determine a consent condition, a mean concentration of 25 mg/l should be achieved.

Standards from other countries

As previously mentioned and discussed in Section 2.3.2, standards for the protection of fresh water fish have been set by the CCREM (1987), the US EPA (1986) and in Alaska (Lloyd 1987). Those set for Canada are only concerned with SS, those set by the US EPA (1986) are concerned with both SS and turbidity, while in Alaska (Lloyd 1987) the standards are only concerned with turbidity.

Discussion

The relationship between turbidity and SS has been discussed in Section 2.3.2. In view of the limited data available for indigenous cyprinids, it is proposed that the same criteria for SS proposed for salmonids fisheries and set by the EC for the protection of EC-designated cyprinid fisheries, that of equal to or less than 25 mg/l SS would be adequate for the protection of indigenous cyprinid fisheries. As with salmonid fisheries, it is not possible, due to limited data, to propose a turbidity standard. However, turbidity standards based on local observations, may provide a quick and easy method to indicate changes that may prove harmful to cyprinid fisheries.

3.4 Biological standards

Compliance with fishery objectives could be assessed by monitoring for biological standards (i.e. making a direct assessment of compliance with use). However, as reported in Section 2.4, a fish classification scheme for use throughout the NRA is currently under development (Mainstone and Wyatt 1991) and biological standards for cyprinid fisheries will not be proposed here.

3.5 Habitat

As mentioned in Section 2.5, it is not within the scope of this report to consider or propose criteria for habitats for cyprinid fisheries. However, habitat alteration may have the effect of drastically altering a fish population regardless of the chemical quality of the water. Thus it is a necessary parameter to consider when regarding the status of a fishery, with both high habitat and chemical quality needing protecting in order to maintain an adequate fishery.

4. MIGRATORY FISHERIES

4.1 Environmental quality objective

The migratory fishery use applies to fresh water (rivers and lakes) and to estuarine and coastal waters. This has changed since Smith *et al* (1991) where the objective only applied to estuarine waters.

EQO

Explanatory note

Maintain water quality so as to protect passage to and from fresh water of all relevant species of migratory fish (where this is not prevented by physical barriers).

To include eels but not marine species which use estuaries for breeding grounds

Migratory fish include salmonids and eels. Eels are generally considered to be more tolerant to poor water quality. Hence when reviewing data, those concerned with salmonids have generally been considered. Migratory Fisheries have not traditionally been considered as a category for EQS derivation and therefore few EQS have been set specifically for this use. Those parameters necessary for maintaining the above objective have been identified from Smith *et al* (1991) and Smith and Gulson (1991). Agg (1989) and the EC Fisheries Directive (CEC 1978) do not consider migratory fisheries. Standards set or proposed for the UK and other countries are reviewed, together with available relevant literature, the List II substances reviewed have been limited to the following:

ammonia, arsenic, boron, chromium, copper, inorganic lead, inorganic tin, iron, mothproofing substances, nickel, organotins, pH, sulphide, vanadium and zinc.

The data for all the considered parameters is critically assessed and where possible, criteria necessary for maintaining the objective are identified or proposed. Proposed standards are summarised in Table 4.1.

4.2 Chemical parameters

4.2.1 Dissolved oxygen for estuarine migration

For the protection of salmonid fisheries, dissolved oxygen requirements need to be adequate for the upstream and downstream migration at least at the respective time of year. Large variations in dissolved oxygen concentrations can occur in estuaries resulting in an 'oxygen-sag', the areas with the highest dissolved oxygen concentrations being the seaward and fresh water points of the estuary. The point of the 'oxygen-sag', where dissolved oxygen is at its lowest, is likely to be the point which blocks migration, however, its position is not fixed and varies with tidal cycles. In addition, the rate of freshwater flow into the estuary as well as the temperature will also affect the dissolved oxygen concentration.

Table 4.1 Environmental quality standards proposed for the protection of migratory fisheries

Parameters	Units	Estuarine/Coastal waters		Status	Fresh water		Status
Dissolved oxygen	mg O ₂ /l	3 (1)	95P	-	4	95P	TG
		5 (1)	AA	-			
Nitrite	µg N/l	- (2)	-	-	270 (3)	95P	-
BOD	mg O ₂ /l	6 (4)	95P	-	6 (4)	95P	-
Chlorine	µg HOCl/l	4	MAX	TG	4	MAX	TG
Ammonia (un-ionized) (total)	µg N/l	21	AA	-	21	95P	-
		780	95P	TG	780	95P	TG
pH (5)	-	6-9	95P	I	6-9	95P	I
List II substances		- (6)	-	-	- (6)	-	-
Temperature (difference) (actual)	°C	<1.5	MAX	-	<1.5	MAX	-
		<21.5	MAX	-	<21.5	MAX	-
Suspended solids	mg/l	- (7)	-	-	- (7)	-	-

- Notes
- (1) For distances >10km or for the maintenance of high quality migratory fisheries higher standards may be required, see Section 4.2.1.
 - (2) Insufficient data to propose a standard.
 - (3) Where chloride concentration varies largely from 10 mg/l standard should be varied according to EIFAC criteria.
 - (4) Measured as BOD₅ (ATU) i.e. oxygen demand without nitrification.
 - (5) Derogation is allowed in the event of exceptional meteorological or geographical conditions.
 - (6) EQSs have been derived for the protection of fresh and salt water life for the following List II substances: arsenic, boron, inorganic tin, iron, mothproofing substances, organotins, sulphide, chromium, copper, lead, nickel, vanadium and zinc. These standards were not derived for the protection of migratory fish, but due to their transitory presence migratory fish should be protected if the EQSs are met. See text for discussion of the standards.
 - (7) Suspended solids are only likely to have a deleterious effect at high concentrations. Standards should be derived locally as necessary.
- AA Annual average.
I Mandatory.
P Percentile.
MAX Maximum allowable concentration.
TG Tentative guideline.

By far the majority of data for migratory fish is concerned with migration through estuaries rather than in fresh waters. Two major reviews, Hugman *et al* (1984) and Stiff *et al* (1990), on the oxygen requirements of migratory fish have been carried out in recent years. Both have proposed standards for migratory fish through estuaries, in addition, a number of standards for various estuaries have been proposed by the relevant authorities. Proposed standards and relevant studies are outlined below.

Standards applicable to the UK

There are no standards that have been adopted by the DoE for dissolved oxygen for the protection of migratory fish through estuaries. However, a number have been proposed.

Davis (1975) as previously mentioned in Section 2.2.1, estimated three protection levels for various populations of fish. The dissolved oxygen concentration that was said to represent almost ideal conditions for anadromous salmonids in sea water was 9.25 mg/l, while 4.5 mg/l would result in the majority of fish being affected due to low levels of oxygen.

EIFAC (Alabaster and Lloyd 1982) proposed that to allow for upstream migration of adult salmonids during the periods of lowest flow, 50 percentile and 95 percentile, in the summer, in areas where dissolved oxygen concentration is at its lowest, should be 5 and 2 mg/l respectively.

- Reviewing data on the dissolved concentration of oxygen in UK estuaries and the migration fish, Hugman *et al* (1984) suggested two standards; a guideline standard of 5 mg DO/l as a lower 95 percentile to protect migratory salmonid fisheries given variation in physical and chemical conditions between various estuaries, and a minimum 95 percentile of 3.0 mg DO/l to permit limited migration of salmonids through estuaries.

Elliot *et al* (1988) mentioned that the fresh-brackish water transition zone of estuaries often has a high turbidity and dissolved oxygen minimum and thus it will be a critical zone for migrating fish. In order to avoid salmonid smolt and grilse kills that had occurred in late spring and summer, due to poor water quality, a depth average dissolved oxygen EQS of 4 mg/l (said to be in practise a target of 4.5 mg/l as a 95 percentile) has been adopted for the upper Forth. The Humber Estuary Committee (HEC) (unpublished) have recently reviewed the EQS for dissolved oxygen in the Humber Estuary with regard to the passage of migratory fish through it. They concluded that a 5 percentile of 5 mg DO/l and an absolute minima of 3 mg DO/l would allow migration to occur.

In his review Hugman *et al* (1984), reported DO standards proposed by different UK regulatory authorities for the maintenance of regular migratory salmonid fisheries in estuaries (Table 4.2). These are in the range 5 to 6 mg/l as minimum annual 95 percentiles.

Table 4.2 Proposed standards for dissolved oxygen concentrations in estuaries for the protection of migratory salmonid fisheries (mg/l) (Hugman *et al* 1984)

Origin	Minimum percentile [oxygen]			Comments
	100%	95%	50%	
Severn Estuary Technical Working Party	-	6	-	samples taken throughout the year
North West Water Authority	3	5	7	samples taken throughout the year
Welsh Water Authority	-	5	-	samples taken throughout the year

The most recent review on the DO requirements of migratory fish was by Stiff *et al* (1990). They proposed a minimum acceptable median and 95 percentile values of 5 mg DO/l and 3 mg DO/l to allow migration of salmonid fish in estuaries but added that higher values may be required where distances greater than 10 km are to be traversed.

Recent data

Data obtained by Hugman *et al* (1984) from Water Authorities and River Purification Boards on dissolved oxygen concentrations and fisheries status for estuaries in the UK indicated that estuaries with minimum mean oxygen concentrations greater than 8.2 mg/l and a minimum lower 95 percentile of at least 4.4 mg/l maintained regular migratory fisheries. Where salmon were beginning to return, (River Thames and Trent) mean oxygen concentrations were above 6.4 mg/l and 3 mg/l for 95 percentile values, although oxygen restrictions sometimes occurred. Hugman *et al* (1984) also included data on stratified estuaries. The Tyne estuary due to stratification was found to have a depth average 95 percentile of only 0.6 mg/l. It was suggested that the ability of the fish to select the depth containing the highest DO allowed for their migration to occur and this being the case the relevant 95 percentile value would be 2.5 mg/l. Additionally data from the Clyde estuary also suggested that salmon smolts are capable of actively avoiding water with low DO concentrations if alternative, better quality water is available.

Stiff *et al* (1990) cited a number of papers that estimated dissolved oxygen requirements of migratory fish over distance.

For adult migration, Curran and Henderson (1988) estimated a DO concentration requirement through a 8 km estuary of 5 mg/l and 2 mg/l as median and 90 percentile values respectively, however, they felt that these may be too stringent. Alternatively, Alabaster and Gough (1986) estimated median and 90 percentile concentrations of 3.8 and 2.2 mg/l for migration over 10 km in the Thames estuary. Comparison of dissolved oxygen

concentrations and expected/actual migration runs, led Alabaster to deduce for distances of 0, 8 and 32 km, median and 95 percentile dissolved oxygen concentrations of 3.6, 4.4, 6.4 mg/l and 2.6, 3.2, 5.4 mg/l respectively would be needed at temperatures between 18.4 and 20.3°C, for migration to occur. Using radio tags attached to *Salmo salar*, Aprahamian *et al* 1988, investigated migration through the Usk estuary, an estuary subject to an oxygen sag due to resuspended sediments, with dissolved oxygen concentration falling to 3 mg/l or below over several kilometres. Only a few fish (29%) migrated upstream at dissolved oxygen concentrations between 3.3 and 4.5 mg/l, at concentrations between 4.7 and 5.2 mg/l, 46% migrated.

For smolt migration Curran and Henderson (1988) stated that seaward migration appeared to be through passive drift during ebb and the smolts may descend up to 10 m to optimise this drift. From data on smolt runs and dissolved oxygen concentrations in the River Gryffe, Clyde and Clyde estuary, Curran and Henderson estimated the 90 percentile threshold DO values for successful smolt migration to be 2 mg/l over 1 - 4 km, 2.5 mg/l over 6 km and 3 mg/l over 8 km. A 90 percentile DO of 1.9 mg/l over 6 km represented an impenetrable barrier to smolt migration.

Alabaster (1988) found chinook salmon, *Oncorhynchus tshawytscha*, preferred dissolved oxygen concentrations of 3.5 mg/l and above for upstream migration.

The only data available since the review by Stiff *et al* (1990) is Alabaster *et al* (1991). They analysed data from the Thames estuary relating the percentage return and the rate of migration of Atlantic salmon (*Salmo salar* L.) to the environmental conditions in the estuary at that time. Monthly catches and percentage returns of grilse in July and September were 'drastically' reduced at 95 percentile concentrations of 2.2 and 2.7 mg/l dissolved oxygen respectively, the weekly return was reduced to zero at 2.4 mg/l (the average flow observed during 1987 - 1988 was 15 m³/s). The authors stated this closely resembled the results obtained by Aprahamian *et al* (1988) for the River Usk.

From the available data, it can be concluded that dissolved oxygen standards for estuaries need to be set with respect to upstream (adult) migration. Downstream smolt migration has been shown to be mainly passive drift and smolts are able to migrate through areas of lower dissolved oxygen than will allow upstream migration. Comparison of the data from studies and proposed standards indicates that the standards proposed by Stiff *et al* (1990), a minimum median and 95 percentile of 5 mg/l and 3 mg/l dissolved oxygen respectively, will be adequate to establish migratory fisheries in estuaries. However, it should be noted that over distances greater than 10 km, higher levels may be needed. Furthermore the maintenance of high quality migratory fisheries may require higher standards such as the 5 mg/l as a 95 percentile suggested by Hugman *et al* (1984) and the Humber Estuary Committee.

A number of studies have indicated that migratory fish are able to actively avoid low concentrations of dissolved oxygen and some authorities have set DO standards for the passage of migratory fish through stratified estuaries as depth average concentrations. However, since the data are limited it is not possible to propose a depth average DO concentration for all stratified estuaries, but instead that the above proposed concentrations be applied to the most oxygenated layer in a stratified estuary. However, a proviso should

also be added to allow lower 95 percentiles when a depth average concentration, shown not to have any deleterious effects on the passage of migratory fish, has been set by various authorities.

4.2.2 Dissolved oxygen for fresh water migration

Standards applicable to the UK

No standards for the protection of salmonids as they migrate through fresh water have been adopted by the DoE however some have been proposed.

As previously stated in Section 2.2.1, EIFAC (Alabaster and Lloyd 1982) and Stiff *et al* (1990) proposed minimum sustained dissolved oxygen concentrations for maintaining the normal attributes of the life-cycle of fish without significant impairment. EIFAC proposed that 5 mg/l dissolved oxygen would be adequate for maintaining cruising swimming speed of fish and upstream migration of Pacific salmon in fresh waters, while Stiff *et al* (1990) proposed that 6 and 5 mg/l would be adequate for maintaining the swimming speed of juvenile and adult salmonids respectively.

Relevant data

The maintenance of swimming performance is important for continued upstream migration. Alabaster and Lloyd (1982) noted fish continued to swim at near lethal levels of dissolved oxygen but maximum sustainable swimming speeds of salmonids normally declined with any reduction below saturation. Smith *et al* (1971) (cited in Alabaster and Lloyd 1982) found adult migrant coho salmon (*Oncorhynchus kisutch*) continued to swim at about 560 mm/s for an hour at dissolved oxygen concentrations of 5 to 6.6 mg/l and for a further hour at concentration of 4.5 to 5 mg/l. However, fish exposed at 4 to 4.5 mg/l for the second hour became fatigued.

As previously mentioned Henderson estimated the 90 percentile threshold DO values for successful smolt migration, the data included observations in rivers as well as estuaries. Threshold values were, 2 mg/l over 1 - 4 km, 2.5 mg/l over 6 km and 3 mg/l over 8 km. A 90 percentile DO of 1.9 mg/l over 6 km represented an impenetrable barrier to smolt migration.

Discussion

The data on migration of adult and smolt salmonids through fresh water are limited and for other species no data is available. At concentrations above 4 mg/l dissolved oxygen, upstream migration appears to occur, however, over long distances at this level the fish may fatigue or a decrease in swimming speed may occur sooner than at higher concentrations. It is therefore suggested that 4 mg/l dissolved oxygen as a 95 percentile be used as a tentative guideline for upstream migration in fresh water. Downstream migration of smolts should be adequately maintained at oxygen levels that allow upstream migration to occur.

4.2.3 Nitrite

Nitrite (NO_2^-) (an intermediate oxidation state between ammonium and nitrate) is a naturally occurring anion both in fresh and marine waters. Its typical concentration in oxygenated waters being less than 0.005 mg/l (Lewis and Morris 1986).

Eddy *et al* 1983 (cited in Lewis and Morris 1986) stated that nitrite ions are actively taken up by most fresh water fish, and that concentrations in the blood plasma may reach ten times that of the surrounding medium. From the blood plasma, nitrite diffuses into red blood cells where it oxidises the iron in haemoglobin to the 3+ oxidation state, forming methaemoglobin, which lacks the capacity to bind oxygen reversibly. Reductase converts methaemoglobin back to haemoglobin but, this is slow, the normal proportion of haemoglobin being restored in 24 - 48 hours on transferral to water lacking nitrite (Huey *et al* 1980, cited in Lewis and Morris 1986).

Standards applicable to the UK

There are currently no water quality criteria set or proposed for the protection of migratory fisheries in the UK. Standards for resident populations of salmonids and cyprinids have previously been discussed in Sections 2.2.2 and 3.2.2.

Standards from other countries

The US EPA (1986) and the CCREM (1987) both have set recommended levels for the protection of salmonid and cyprinid fish, (see Sections 2.2.2 and 3.2.2) however it is possible that these criteria which appear to be designed for the protection of a resident population may be too strict for a migratory population.

Recent data

Crawford and Allen (1977) using static systems found in fresh water mortality of chinook salmon fingerlings only occurred after 48 hours exposure at 10 mg NO_2^-/l , but stress was observed at 0.5 mg NO_2^-/l . The fish responded to nitrite poisoning by decreasing their activity and resting motionless at the bottom.

Lewis and Morris (1986) reported that nitrite toxicity is exacerbated by low dissolved oxygen because of the reduction in the oxygen capacity of the blood. Inactive fish would have a low oxygen demand and therefore may not be immediately threatened by severe methaemoglobinemia. However, for upstream migrating fish, which have a large oxygen demand, this may affect their passage, while smolts migration downstream, which is by a mainly passive process, would be less affected by low oxygen concentration. In addition, high temperatures, which reduce the dissolved oxygen, may exacerbate the effect on upstream migration.

In salt water, Crawford and Allen (1977) found the toxicity of nitrite to chinook salmon fingerlings (*Oncorhynchus tshawytscha*) decreased. In fresh water, the 48 hour LC50 was 19 mg/l nitrite while in natural sea water, 48 hour 10% mortality occurred at 1070 mg/l nitrite, it was not reported if this was also for older migrating salmon.

Discussion

Data on the toxicity of nitrite to migratory fish are very limited, however it does appear that while toxicity may decrease in salt or estuarine waters the effect of nitrite on upstream migratory fish may be exacerbated at low oxygen concentrations.

Considering the reduction in nitrite toxicity in salt water, Smith *et al* (1991) tentatively proposed the EIFAC (1984) standard for nitrite in the presence of 40 mg Cl/l and 0.45 mg NO₂-N/l, as a 95 percentile would be adequate for the protection of migratory fish through marine and estuarine waters. However, data for nitrite toxicity in these waters are very limited and it is not possible to propose an EQS for the protection of migratory fish in estuaries and marine waters.

For fresh waters, it is proposed that the standards proposed for salmonid resident population, 0.27 mg NO₂-N/l as a 95 percentile, should protect migratory fisheries. As with salmonid fisheries (Section 2.2.2) if chloride concentrations vary widely from 10 mg/l on which this standard is based, the standard should be varied according to the EIFAC criteria.

4.2.4 Biochemical oxygen demand

There appear to be no standards for BOD for the protection of migratory fisheries. Dissolved oxygen requirements proposed in this report for migratory fisheries (Section 4.2.1) are similar to those proposed for cyprinids (Section 3.2.1) rather than for salmonid fisheries (Section 2.2.1). In the absence of any data on BOD and migratory fisheries, it is proposed that the BOD standard proposed for cyprinid fisheries (Section 3.2.3), 6 mg O₂/l as BOD₅ (ATU), i.e. oxygen demand without nitrification, be used as a guideline along with a proviso allowing higher levels as long as dissolved oxygen concentrations are not reduced below the relevant criteria needed for protection of migratory fish.

4.2.5 Phosphorus

Standards for phosphate in order to protect against eutrophication have been set in the Fisheries Directive (CEC 1978). However, as previously mentioned (Sections 2.2.4 and 3.2.4) phosphate itself is not directly toxic to fish, rather effects are due to a reduced oxygen concentration that may occur due to algal blooms. It is therefore concluded that since detriments to other water uses are likely to occur due to eutrophication before effects on fisheries, a standard for the protection of migratory fisheries is not necessary.

4.2.6 Chlorine

In water chlorine undergoes a pH-dependent reaction to produce HOCl which together with Cl₂ comprises 'free chlorine'. Additional reactions with amines and ammonia result in the formation of chloramines or 'combined chlorine'. Thus the 'total (residual, available) chlorine' comprises 'free' and 'combined' chlorine. EIFAC (Alabaster and Lloyd 1982) considered that the toxic chemical species in solution in chlorine poisoning is hypochlorous acid (HOCl) and thus the toxicity of any solution containing chlorine will depend on the pH of the solution as this will markedly influence the degree of dissociation.

Standards applicable to the UK

There are no UK mandatory standards for chlorine for the protection of fresh, estuarine or coastal migrating fish. However, standards have been set by the EC (CEC 1978) for the protection of EC-designated salmonid and cyprinid fisheries and proposed by EIFAC (Alabaster and Lloyd 1982). These have been outlined in Sections 2.2.5 and 3.2.5. In addition, Hedgecott (1990) proposed an EQS of 5 µg/l total chlorine for the Humber estuary as a 95 percentile for protection of aquatic organisms in the estuary.

Standards from other countries

There appear to be no standards for chlorine from other countries for the protection of migratory fisheries. Criteria for the protection of fresh water aquatic life have been proposed by the CCREM (1987) and the US EPA (1985), these have been discussed in Sections 2.2.5 and 3.2.5. In addition, the US EPA (1986) have set a standard for chlorine-produced oxidants for the protection of salt water life, of 7.5 µg/l as a four day average and 13 µg/l as a one hour average, which can be exceeded once every three years.

Discussion

Hedgecott (1990) reviewing toxicity data for salt water organisms and field data concluded that a concentration of 5 µg/l total chlorine expressed as a 95 percentile would be acceptable for the protection of aquatic life in the Humber Estuary, this included the passage of migratory fish. A maximum acceptable concentration of 10 µg/l was also proposed. Hedgecott (1990) cited Realey (1989) who found active avoidance by *Oncorhynchus kistutch* of 2 µg/l chlorine. However, it is probable that active avoidance at such low concentrations would give protection from intermittent discharges, and would be unlikely to prevent migration through an estuary where there are likely to be areas of lower concentration.

Alabaster and Lloyd (1982) reported that acutely lethal levels of chlorine are little changed by increasing salinity up to 50% sea water.

Based on the previously reviewed criteria (Section 2.2.5) and the additional data on toxicity in salt water and migratory fish, it appears that the criterion, 4 µg/l HOCl (maximum allowable concentration), as proposed for salmonid and cyprinid fisheries, if applied to estuaries and fresh water, would not inhibit the migration of fish through them. However, this can only be tentatively proposed as information on migratory fish in these reviews has been very limited.

4.2.7 Ammonia

Standards applicable to the UK

Limited data are available for ammonia toxicity to migratory fish and no water quality standards specifically for migratory fish have been set. Seager *et al* (1988) proposed an EQS of 0.021 mg NH₃-N/l expressed as annual average of un-ionized ammonia, for the protection of marine fish, however it was not stated if they thought this would be adequate to protect the migration of salmonids through estuarine and marine waters. Standards proposed for

resident fresh water salmonid and cyprinid populations have previously been outlined and discussed in Sections 2.2.6 and 3.2.6.

Discussion

Since Seager *et al* (1988), there appear to be no data on the effect of un-ionized ammonia on migratory fisheries. However, Seager *et al* (1988) cited a number of studies, which are outlined below which are of relevance to the passage of migratory fish.

Seager *et al* (1988) concluded that the acute toxicity of un-ionized ammonia decreases with increasing salinity to a minimum point (corresponding to approximately 10 ppt or 30% sea water) and then increases, to a level similar to that observed in fresh water, as the salinity approaches that of full-strength sea water. Citing Alabaster *et al* (1979) and Alabaster *et al* (1983), Seager *et al* (1988) indicated that while the toxicity of un-ionized ammonia decreased with increasing salinity, it increases at low dissolved oxygen concentrations, but acclimation to low dissolved oxygen concentrations may again decrease toxicity. At low dissolved oxygen, (approximately 3 mg/l) at 0 ppt, LC50s of un-ionized ammonia to Atlantic salmon smolts were in the range 0.06 - 0.07 mg NH₃-N/l, after 24 hour acclimation to the dissolved oxygen concentration, the toxicity had decreased such that the LC50 was 0.12 mg NH₃-N/l. At a higher dissolved oxygen concentration, (approximately 9 - 10 mg/l) DO, and 0 ppt the LC50 further decreased to 0.12 - 0.17 mg NH₃-N/l and after 24 hour acclimation the LC50 was 0.16 mg NH₃-N/l.

At low dissolved oxygen (approximately 3 mg/l) but higher salinity, 10 ppt, the LC50 was in the range 0.09 - 0.12 mg NH₃-N/l, and decreased to 0.15 mg NH₃-N/l after 24 hour acclimation to the low dissolved oxygen concentration. At a higher dissolved oxygen concentration (approximately 9 - 10 mg/l) the LC50 was in the range 0.21 - 0.23 mg NH₃-N/l, and decreased to 0.41 mg NH₃-N/l after 24 hours acclimation to DO level.

Observations, cited in Seager *et al* (1988), in the Mersey estuary, indicated a concentration of 0.008 mg NH₃-N/l was passable for salmonids but a concentration of 0.29 mg NH₃-N/l was impassable for flounder.

The toxicity data on the effects of ammonia to migratory fish are limited but toxicity does appear to decrease with increasing salinity, until a certain salinity is reached where it increases to levels similar to those found for fresh water. This decrease in toxicity may provide some protection for migratory fish within estuaries, especially in areas where an oxygen sag may occur. Because of the effect of acclimation to low dissolved oxygen reducing the toxicity of un-ionized ammonia, Seager *et al* (1988) concluded that there may be a slight increase of safety for smolts as they migrate seaward in addition to protection gained from dilution.

There is insufficient data to propose specific standards for the protection of migratory fisheries. However, as previously stated, 0.021 mg NH₃-N/l (annual average) was proposed by Seager *et al* (1988) for the protection of marine fish and this value was also proposed in Section 2.2.6 as a 95 percentile for the protection of resident salmonid fisheries. It is therefore proposed, that these standards, applied in salt and fresh waters respectively, should give adequate protection for the passage of migratory fish.

As mentioned in Section 2.2.6, Smith and Gulson (1991) discussed the need for standards for total ammonia to provide a firm basis for the calculation of discharge consents. They concluded that due to differences in ambient pH and temperature characteristics between rivers further research was needed before a meaningful standard for total ammonia, which takes into account these factors in the receiving water, could be set. They suggested that for the protection of fisheries the mandatory standards for EC-designated fisheries should be used as guideline standards. These have been proposed for the protection of salmonid and cyprinid fisheries (Sections 2.2.6 and 3.2.6.) however the Fish Directive (1978) has not designated migratory fisheries and hence no standards for total ammonia are proposed. There are insufficient data on migratory fish to determine a separate standard for total ammonia or to determine if the standards proposed for EC-designated fisheries, 1 mg NH₄/l (equivalent to 0.780 mg N/l), would be too or insufficiently stringent for the protection of migratory standards. In the absence of further information it is therefore proposed that the standards be used as a guideline for total ammonia concentrations for the protection of migratory fish, with the added proviso that higher values should be allowed if shown not to be deleterious to the population.

4.2.8 Arsenic

Introduction

Arsenic may be released into water from mining, smelting and refining and other process either directly or indirectly. Arsenic is also used as a wood preservative, in the manufacture of glass, alloys or medicines.

Standards applicable to the UK

Mance *et al* (1984a) were unable to propose an EQS for anadromous migratory fish through fresh water. They stated that the value proposed for resident populations may be too strict, and recommended that any standards for the passage of such fish should be based on local judgement since other factors such as low flows or DO were normally likely to be limiting to migration.

Mance *et al* (1984a) did propose an annual average EQS of 50 µg/l and 25 µg/l dissolved arsenic for the protection of fresh and salt water fish respectively, these have since been adopted by the Department of the Environment (DoE 1989). However, when proposed, Mance *et al* (1984a) stated that the EQS for salt water fish was extremely tentative due to lack of toxicity data, and it should be reviewed when additional data became available.

Recent studies

Only one study on the effects of arsenic to migratory fish appears to be available. Nichols *et al* (1984) investigated the effects of 6 months exposure of coho salmon, during parr-smolt transformation, to arsenic trioxide. The threshold level for affecting parr-smolt transformation was 0.3 mg As/l. This significantly reduced successful seaward migration and had transient effects on physiological parameters.

Discussion

The only available data on the effects of arsenic on migratory fish indicate that the threshold level affecting parr-smolt transformation is a factor of ten higher than the EQS proposed for resident populations. Thus, while the available data do not enable a separate EQS to be proposed, the current EQSs for fresh and salt water fish (DoE 1989) do not appear to be too lax to protect migratory fish. Therefore it is proposed that these EQSs will provide adequate protection for this use.

4.2.9 Boron

Standards applicable to the UK

There are no current water quality standards for the protection of migratory fisheries for the UK. Mance *et al* (1988a) were unable to propose a separate standard for boron for the protection of fresh water migratory fish. They stated that the proposed EQS for the protection of resident populations of fresh water fish (2 mg/l, annual average total concentration) (Section 2.2.8) may be too stringent, and any standard for migratory fish should be based on local observations, since other factors such as dissolved oxygen are likely to be limiting.

An EQS of 7 mg/l as an annual average for total boron for the protection of salt water fish and shellfish was proposed by Mance *et al* (1988a). Both EQSs have since been adopted by the DOE (1989).

Discussion

There are no additional data on the toxicity of boron to migratory fish since Mance *et al* (1988a). They were unable to propose EQSs for migratory fish. The EQSs proposed for salt water fish were very tentative, the authors stating that they should be reviewed when additional data became available. The standard was based on very limited data that included toxicity data on the indigenous estuarine species *Limanda limanda* (dab) and the non-indigenous salmonid species coho salmon. However, the reported tests involved LC50 data and thus the relevance of these tests and the standard to fish migrating through coastal and estuarine waters cannot be easily assessed.

Because of the lack of data on the effects of boron on migratory fish, it is not possible to set an EQS specifically for migratory fisheries. However, the EQSs for fresh and salt waters should be adequate to protect such fisheries.

4.2.10 Chromium

Standards applicable to the UK

There are currently no standards adopted or proposed in the UK for the protection of migratory fish in fresh, estuarine or marine waters. Mance *et al* (1984b) stated it was not possible to propose a standard for fresh water through which anadromous migratory fish pass, and that the standard proposed for resident salmonids may be too stringent. They recommended standards should be based on local situations. EQSs for the protection of

resident populations of fresh water fish have been proposed, these have been previously outlined in Sections 2.2.9 and 3.2.9. In addition, Mance *et al* (1984b) proposed an EQS of 15 µg Cr VI/l as an annual average dissolved concentration for the protection of salt water life. All these EQSs have since been adopted as total chromium (III and VI) by the DoE (1989).

Discussion

No new data on the effects of chromium on migratory fish, since Mance *et al* (1984b), appear available, and thus it is not possible to set specific standards for this use. However, the EQSs for fresh (for salmonids) and salt waters should adequately protect migratory fisheries.

4.2.11 Copper

Standards applicable to the UK

There are currently no standards for copper set or proposed in the UK for the protection of migratory fisheries either in fresh or salt waters. Mance *et al* (1984c) were unable to recommend a separate standard for migratory fisheries in fresh water and proposed that local conditions should be considered. However, they did propose EQSs for copper in fresh water in the range 1 - 28 µg/l depending on water hardness (annual average dissolved concentration), for the protection of resident populations of fish and an EQS of 5 µg/l (annual average dissolved concentration) for the protection of salt water life. These have since been adopted by the DoE (1989).

Discussion

Mance *et al* (1984c) cited no specific data concerning the migration of fish and no data are available in the recent literature. In the absence of data it is not possible to propose a separate standard for copper for the protection of migratory fisheries. However, the EQSs for fresh (for salmonids) and salt waters should be adequate to protect migratory fisheries.

4.2.12 Inorganic lead

Standards applicable to the UK

Brown *et al* (1984) were unable to propose an EQS for migratory fish in fresh waters. They suggested that the EQSs for resident populations of salmonids (see Section 2.2.11), which are in the range 4 - 20 µg/l, depending on water hardness, may be too strict and instead standards should be based on local observations.

Brown *et al* (1984) also proposed an EQS of 25 µg/l (annual average) dissolved concentration inorganic lead for the protection of salt water fish, this has been adopted by the DoE (1989) for the protection of salt water life.

Discussion

Since the review by Brown *et al* (1984) there appear to be no additional data on the effects of inorganic lead to migratory fish.

While Brown *et al* (1984) proposed standards for the protection of salt water fish, of the data reviewed none was for salmonid species, so the adequacy of this standard for the protection of estuarine or coastal water for migratory fish cannot be assessed.

Examination of data for lead in UK rivers shows that annual average concentrations of dissolved lead seldom exceed 10 µg Pb/l and there is no evidence of lead concentrations restricting the development of fish populations (Brown *et al* 1984). In addition, the concentrations of lead in European estuarine waters were in the range 0.1 to 170 µg/l and 0.03 to 58 µg/l for European coastal waters.

Because of the lack of appropriate data for migratory fish no standards can be set specifically for the protection of migratory fish. However, the mandatory EQS values for fresh (for salmonids) and salt waters (DoE 1989) will provide adequate protection for the use.

4.2.13 Inorganic tin

Standards applicable to the UK

There are currently no environmental quality standards proposed for the protection of migratory fish in fresh or marine waters.

Mance *et al* (1988b) stated that they were unable to propose a standard for anadromous fish, and that the standard proposed for resident fish populations was likely to be too stringent. They suggested that standards should be based on local observations and other factors, such as low dissolved oxygen, were likely to be limiting to migration first.

Tentative EQSs of 25 µg/l and 10 µg/l (annual average dissolved concentration) have been proposed by Mance *et al* (1988b) for the protection of fresh and salt water resident populations of fish. However, they have yet to be adopted as national EQS values. The only available toxicity data on marine fish were limited to a four day study on *Limanda limanda* which showed no mortality at 1 mg Sn/l (Hugman and Mance 1983).

Discussion

Since the review by Mance *et al* (1988b), the only additional data on marine fish were acute toxicity tests carried out by Taylor *et al* (1985) on *Limanda limanda* (dab). These showed the 96 hour LC50 to be over 0.035 mg Sn/l.

Mance *et al* (1988b) reported that data on concentrations of inorganic tin in fresh, estuarine and coastal waters was basically restricted to North America, but environmental concentrations appeared to be limited to concentrations of nanograms/l.

No additional data on the effects of inorganic tin on migratory fish in fresh, estuarine and marine waters have been reported and thus it is not possible to propose standards specifically for this use. However, EQS for fresh and salt waters should adequately protect this use.

4.2.14 Iron

Standards applicable to the UK

No standards for iron for the protection of migratory fish have been proposed in the UK.

An EQS of 1 mg Fe/l (annual average dissolved concentration) has been adopted by the DoE (1989) for the protection of fresh water and salt water life. This is stricter than the original EQSs, 2 mg Fe/l and 10 - 25 mg Fe/l (annual average total concentration) proposed by Mance and Campbell (1988) for fresh water and salt water fish respectively.

New and relevant data

Mance and Campbell (1988) did not present any toxicity data that may be related to migratory fish and none since appear to be available.

For estuaries, Mance and Campbell (1988) did not find any studies directly linking observed concentrations of iron with biological status. However, using the classification schemes of the 1980 River Survey, they found concentrations as high as 21 mg/l would be acceptable in an estuary with a large tidal range. Therefore they recommended a range of EQSs for total iron in estuaries, the allowable concentration depending on the tidal current and concentration of suspended solids. However, they also recommend, based on field observations, that the annual average concentration of dissolved iron should not exceed 1 mg/l as an annual average.

The lack of information on the effects of iron on migratory fisheries makes it difficult to propose standards specifically for their protection. However, data on fresh water and marine fish have shown that even at high concentrations reasonable fisheries can exist. Thus it is proposed that EQSs adopted by the DoE (1989) for fresh and salt waters should adequately protect the passage of migratory fish.

4.2.15 Mothproofing substances

Standards applicable to the UK

There are currently no standards for mothproofing substances specifically for the protection of migratory fisheries. EQS have been proposed by Zabel *et al* (1988a) for four mothproofing chemicals for the protection of fresh water fish and these have since been adopted by the DoE (see Section 2.2.14). In addition, Zabel *et al* (1988a) proposed an EQS for permethrin of 0.001 µg/l (total active ingredient) as a 95 percentile for the protection of salt water fish, however, the EQS proposed by Zabel *et al* (1988a) for the protection of fresh water fish were also adopted by the DoE (1989) for the protection of salt water life. There appears to be no information on the effects of mothproofing substances to migratory fisheries. However, it is proposed that the EQS for the protection of fresh and salt water should be adequate for the protection of migratory fisheries.

4.2.16 Nickel

Standards applicable to the UK

Mance and Yates (1984a) did not propose a standard for migratory fish in fresh water, stating that the EQS proposed for the protection of fresh water fish, 50 - 200 µg Ni/l (annual average dissolved concentration, depending on water hardness) may be too stringent and that standards should be related to local conditions.

An EQS for the protection of salt water life of 30 µg Ni/l (annual average dissolved concentration) was also proposed.

Discussion

There appears to be no additional data on the effect of nickel on migratory fish since Mance and Yates (1984a), and what was presented was extremely limited. Thus while the lack of data makes it impossible to propose standards specifically for migratory fisheries, the EQSs for fresh and salt waters should adequately protect such fisheries.

4.2.17 Organotins

Standards applicable to the UK

No standards for organotins have been specifically proposed in the UK for the protection of migratory fisheries. As mentioned in Section 2.2.16, Zabel *et al* (1988b) have proposed EQSs of 20 ng/l (total) as 95 percentiles for tributyltin (TBT) and triphenyltin (TPT) which have since been adopted by the DoE (1989) as maximum allowable concentrations.

Zabel *et al* (1988b) also proposed EQSs of 1 and 8 ng/l (total) again as a 95 percentiles for TBT and TPT respectively, for the protection of salt water fish. Since then the DoE (1989) have adopted EQSs of 2 and 8 ng/l (total) as a maximum allowable concentration for the protection of salt water life.

Discussion

As previously mentioned in Sections 2.2.16 and 3.2.16 the EQSs for TBT and TPT for the protection of fresh water fish are tentative. Those proposed by Zabel *et al* (1988b) for the protection of salt water fish are also tentative. As for salmonid and cyprinid fisheries, more data, which are currently not available, are needed before a standard specifically for the protection of migratory fisheries can be proposed. However, the EQSs for fresh and salt water (DoE 1989), which must be applied to discharges containing these substances should be adequate to allow the passage of migratory fish.

4.2.18 pH

Standards applicable to the UK

There appear to be no standards for pH, either in the UK or other countries, specifically for the protection of migratory fisheries.

Wolff *et al* (1988) recently reviewed data on the effects of pH on fresh water fish and proposed an EQS of pH 6.0 - 9.0 (95 percentile) (since adopted by the DoE 1989) for non-EC salmonid and cyprinid fisheries. Standards have also been set by the EC for EC-designated salmonid and cyprinid fisheries, these have been outlined in Sections 2.2.15 and 3.2.15. However, Wolff *et al* (1988) were unable to propose a standard for migratory fish in fresh water, and stated that the standard proposed for resident populations of fresh water fish may be too stringent. They suggested that standards should be based on local observations, since other factors as well as the time taken to travel through a stretch of water may have an effect.

Wolff *et al* (1988) also proposed an EQS of pH 6.0 - 8.5 (95 percentile) for the protection of salt water fish, this also has since been adopted by the DoE (1989). However, its relevance to the passage of migratory fish is difficult to assess. This pH range has also been set by the US EPA (1986) for the protection of marine aquatic life, but they state that the pH should not change by more than 0.2 units outside of the normally occurring range.

Due to the large concentration of salts present the buffering capacity of marine waters is far greater than that of fresh waters, and thus the naturally occurring variation of pH is less than in fresh water. Capurro (1970, cited in US EPA 1986) stated the normal pH values in sea water are 8.0 to 8.2 at the surface, decreasing to 7.7 to 7.8 with increasing depth.

Discussion

A number of studies (see Section 2.2.17) have indicated that the decline of fresh water fisheries may be due to episodic kills of fish, rather than massive fish kills. These acid episodes, (flushes of acid water) are most common during periods of heavy rain and rapid runoff when there is little opportunity for the buffering effect of any mineral soil to occur. EIFAC (Alabaster and Lloyd 1982) concluded that the threshold pH for salmonid parr was pH 5.3, while MAFF (1991) indicated that at the time of migration salmon smolts, due to physiological changes may be particularly vulnerable to acid conditions.

Since Wolff *et al* (1988) there have been no additional data on the effects of pH on migratory fisheries. While it appears that migratory fish, at least salmonids, may be more susceptible to episodic flushes of low pH, there is no data to indicate that migratory fish will not be protected by the EQS of pH 6 - 9 proposed for salmonid and cyprinid resident fisheries (see Sections 2.2.17 and 3.2.17) in fresh waters or by a range of pH 6 - 8.5 in salt water.

It is not possible to propose an EQS for pH specifically for the migration of fish through fresh, estuarine or marine water due to the limited data. However, the national EQS pH values for fresh and salt water (DoE 1989) should provide adequate protection along the migratory route.

4.2.19 Sulphide

Standards applicable to the UK

No standards have been specifically proposed in the UK for sulphide for the protection of migratory fisheries. Mance *et al* (1988c) have proposed an EQS for the protection of fresh water fish (see Section 2.2.18), and an EQS of 10 µg/l (undissociated hydrogen sulphide) as a 24 hour maximum average concentration for the protection of salt water life was also proposed. None of these EQSs have yet been adopted by the DoE.

Discussion

The EQSs for the protection of fresh and salt water fish proposed by Mance *et al* (1988c) are tentative, with the EQS for the protection of salt water fish being based on data for salt water organisms other than fish for which no information was available. The lack of data on migratory fish in Mance *et al* (1988c) make it impossible for an EQS specifically for the protection of migratory fisheries to be proposed. However, the proposed EQSs should afford adequate protection of migratory fisheries. The EQSs for sulphide are currently being reviewed to include more recent data and to derive a more practicable statistical expression for the standard.

4.2.20 Vanadium

Standards applicable to the UK

No environmental quality standards for vanadium have been specifically proposed for the protection of migratory fish. Mance *et al* (1988d) stated that available information was not sufficient to propose a separate standard for the protection of fresh water migratory fish and the EQS proposed for a resident fresh water salmonid population may be too stringent. They suggested that any standard should take into account local conditions since factors other than vanadium are likely to initially be limiting to migration.

Mance *et al* (1988d) proposed EQSs for total vanadium, for the protection of both fresh and salt water fish and these have since been adopted by the Department of the Environment (DoE 1989). Standards for the protection of fresh water fish, 20 or 60 µg/l vanadium, depending on water hardness (annual average concentration), have been previously outlined in Section 2.2.19. The standard proposed for the protection of salt water fish is an annual average concentration of 100 µg/l total vanadium.

Discussion

Mance *et al* (1988d) believed that the EQSs for vanadium proposed for the protection of resident fresh water fish would be too stringent for application to migratory fish. In addition, the EQS adopted for the protection of salt water fish (DoE 1989) used data on only one fish, *Limanda limanda* (dab) in the derivation, and thus its relevance to migratory fish moving through estuaries cannot be adequately assessed.

A review of data has revealed only one recent study that may be relevant to migratory fish. Hamilton and Buhl (1990) reported 24 and 96 hour LC50s of 46.5 mg/l and 16.5 mg/l

respectively in reconstituted brackish water (343 mg/l CaCO₃, pH 7.4 - 8.3) for chinook salmon. These values fall within the range of toxicities reported by Mance *et al* (1988d) for fresh water fish and thus, at least for the non-indigenous chinook salmon, the toxicity of vanadium does not appear to increase with increasing salinity.

Because of the paucity of data it is not possible to propose a specific standard for the protection of migratory fisheries. However, the EQSs for fresh and salt waters should be adequate to protect this use.

4.2.21 Zinc

Standards applicable to the UK

No standards for the protection of migratory fish in fresh, estuarine or marine waters have been proposed or adopted.

Mance and Yates (1984b) were unable to propose an EQS for fresh water migratory fish, they suggested that the EQS for resident populations of salmonids may be too stringent to be applied for the passage of migratory fish. They recommended that any EQS standard for the passage of migratory fish should be based on local observations, since other factors such as dissolved oxygen concentrations or low flow are normally the critical constraint.

Mance and Yates (1984b) have proposed EQSs for resident populations of fresh water fish, these vary with respect to water hardness and have been previously outlined in Sections 2.2.20 and 3.2.20. In addition, they proposed an EQS of 40 µg/l total zinc (annual average concentration) for the protection of marine fish. All these EQSs have since been adopted by the DoE (1989). The data on migratory fish used in the derivation of the salt water EQS were limited to 48 hour LC50s for *Salmo salar* smolts, thus it is difficult to assess its application to the passage of migratory fish.

Discussion

Alabaster and Lloyd (1982) noted that the toxicity of zinc was increased by a decrease in dissolved oxygen content and decreased by an increase in salinity. Thus an increase in toxicity of zinc to smolts or upstream migrating fish in estuaries where there is an oxygen sag, may be counteracted by the effects of salinity. However, there appear to be no available toxicity data since Mance and Yates (1984b) on the effects of zinc to migratory fish and thus it is not possible to propose specific standards for the protection of migratory fisheries. Since the available data, do not show that EQSs for fresh (for salmonids) and marine waters be too lax, it is proposed that these should be adequate for the protection of this use.

4.3 Physical parameters

4.3.1 Temperature

The tolerance of fresh water fish to high temperature depends on a number of factors including the species, stage of development, season, acclimation temperature, concentrations of dissolved oxygen and the presence of pollutants. Changes in the natural temperature régime may produce changes in the population composition, and fish behaviour.

Standards applicable to the UK

No standards for temperature for the protection of migratory fish have been adopted in the UK, however, the Humber Estuary Committee (HEC) have recently proposed a temperature standard for the estuary with regard to the migration of fish through it (HEC, unpublished 1991). They decided in order to allow the migration of fish through the estuary, a maximum temperature of 24°C was suggested but that 21.5°C should be used as a 'working rule of thumb'.

Standards from other countries

In order to assure protection of the characteristic indigenous marine community of a water body segment from adverse thermal effects, the US EPA (1986) have set temperature criteria for marine waters. These apply to estuaries, and so should allow for the passage of migratory fish, and are also designed to prevent alteration of natural temperature cycles, which may be involved with stimulation of upstream migration, etc. The criteria are:

1. The maximum acceptable increase in the weekly average temperature resulting from artificial sources is 1°C during all seasons of the year providing that the summer maxima are not exceeded.
2. Daily temperature cycle characteristic of the water body segment should not be altered in either amplitude or frequency.

Summer thermal maxima that define the upper thermal limits for the communities of the discharge area should be established on a site specific basis.

Recent data for upstream (adult) migration

Alabaster (1990) investigated the migration temperature requirements of Atlantic salmon (*Salmo salar*) by analysing data on their upstream migration in the River Dee, Wales. When discounting relationships with the flow, they found the greatest reduction in the rate of migration (60%) corresponded to the highest weekly mean maximum observed water temperature of 21.5°C. Taking into account the effects of flow, the corresponding reduction in proportion counted would be 60% at 10 m³/s and 49% at 15 m³/s. Thus the highest reduction in migration would occur at the highest temperature and the lowest flows. However, the data given are unclear as to where (estuary or river) the temperatures were monitored.

Extrapolating the data to investigate the effects of further increase in temperature on the run of the fish, Alabaster showed that if flow was neglected, the rate of migration was estimated to be reduced to about half of the average at mean weekly maximum water temperature of 19.5°C and to about a quarter of the average at 25.5°C. Taking the effects of flow into account, the estimated proportional number of fish is reduced to 0 at a mean weekly minimum temperature of 22.1°C for a flow of 10 m³/s and 24.9°C for a flow of 15 m³/s. The author stated that the results were in accord with other studies, but direct comparisons could not be made.

Alabaster *et al* (1971) (cited in Alabaster 1990) reported that the numbers of salmon grilse in Miramichi River, Canada, moving into a trap, decreased with increasing water temperature. Fish were intercepted until the minimum temperature reached 22.5°C (mean 24°C). Also cited in Alabaster (1990), Alabaster (1989) reported that migration of salmon in a river in Devon was reduced with increasing water temperature and that extrapolation of the data found that it would have ceased at monthly mean value of 24.9°C.

In the Thames estuary, the maximum water temperature associated with complete inhibition of the monthly and annual return runs of Atlantic salmon in July and September was 24.2°C and a 50 percentile of 21.5°C was associated with a zero monthly return (Alabaster *et al* 1991).

Jackson and Howie 1967, (Alabaster *et al* 1991) found good runs of salmon take place in the River Erne at temperatures up to 20°C, while Cuinat and Rohuet (Alabaster *et al* 1991) found that in the River Dordogne and River Garonne migration stopped at 25°C.

Wilson *et al* (1987) reviewed the available literature to investigate the effect of temperature on five species of salmon; *Oncorhynchus keta* (chum), *O. kisutch* (coho), *O. gorbuscha* (pink), *O. nerka* (sockeye), *O. tshawytscha* (chinook), commonly found in the Susitna drainage, Alaska. Using data from the literature as well as observed effects in the Susitna drainage, they estimated tolerance and preferred criteria for different life-stages for the five salmonid species (see Table 4.3). The range of temperatures at which adult migration occurred cited in the literature was 2.0 - 21.0°C, while estimated tolerated temperatures for upstream (adult migration) were 1.5 - 18°C, and estimated preferred range was 6.0 - 13°C.

Table 4.3 Estimated temperature (°C) ranges for adult salmonid and smolt migration (Wilson *et al* 1987)

Species/life-stage	Range from cited literature	Estimated tolerance	Estimated preferred
<i>Oncorhynchus keta</i>			
Adult migration	4.4 - 21.0	1.5 - 18.0	6.0 - 13.0
Smolt migration	3.0 - 14.5	3.0 - 13.0	5.0 - 12.0
<i>O. nerka</i>			
Adult migration	5.8 - 15.6	2.5 - 16.0	6.0 - 12.0
Smolt migration	4.0 - 17.8	4.0 - 18.0	5.0 - 12.0
<i>O. gorbuscha</i>			
Adult migration	7.2 - 15.5	5.0 - 18.0	7.0 - 13.0
Smolt migration	4.0 - 14.5	4.0 - 13.0	5.0 - 12.0
<i>O. tshawytscha</i>			
Adult migration	2.0 - 15.6	2.0 - 16.0	7.0 - 13.0
Smolt migration	4.3 - 16.0	4.0 - 16.0	7.0 - 14.0
<i>O. kisutch</i>			
Adult migration	2.0 - 15.5	2.0 - 18.0	6.0 - 11.0
Smolt migration	2.0 - 15.0	2.0 - 16.0	6.0 - 12.0

Recent data for downstream (smolt) migration

Compared to data on adult upstream migration, those on temperature affecting downstream (smolt) migration are limited to non-indigenous species of salmon.

Wilson *et al* (1987) reviewed the available literature to investigate the effect of temperature on five species of Pacific salmon, *Oncorhynchus spp.* see Table 4.3. Estimated tolerance limits for smolt migration were in the range 2 - 18°C, while estimated preferred temperatures were in the range 5 - 14°C.

Crisp (1987) cited Solomon (1978) who found that seaward movement of smolts is inhibited at temperatures around 11°C or below (i.e. smolt migration does not appear to occur until a temperature of around 11°C is reached).

While in laboratory studies, Gray (1990) found juvenile chinook salmon avoided thermal discharges in the laboratory when the increase was 9 to 11°C above ambient, the mean plume temperature causing avoidance, increased with acclimation temperature.

Discussion

A number of studies on the effects of temperature on upstream migration of salmonids have been carried out. However, it is difficult to compare these studies since migration will be affected by a number of factors such as flow, dissolved oxygen concentration and distance travelled as well as temperature which has been measured and reported in different ways. Data on smolt migration have been limited to studies on non-indigenous fish and so it is difficult to assess temperature affects for indigenous species.

The data indicate that good upstream salmonid runs will occur up to about 20°C, even at low flows and dissolved oxygen concentrations. The HEC (1991, unpublished) in reviewing temperature criteria for the Humber Estuary with regard to migrating fish, concluded that a temperature of 21.5°C would allow migration to occur through the estuary. For the majority of the studies here, runs would still occur up to this temperature, although they may be reduced.

Because of the difficulties of setting a temperature standard for migratory fish, the passage of which are also dependent on other factors, it is proposed that it be set as a guideline standard. It is therefore proposed that a guideline maximum standard of 21.5°C be set for the protection of migratory fisheries with the proviso that higher temperatures may be allowed if found not to be detrimental to the passage of fish. This standard should be applied where waters are affected by thermal discharges: natural temperatures may exceed this value. It is also proposed that the standard in the EC Fish Directive (CEC 1978) for salmonid fisheries for temperature difference be applied to discharges in the passage of migratory fish. That is, that the temperature difference at the edge of the mixing zone compared to unaffected water should not exceed 1.5°C, and as in Section 2.3.1 it should be applied as a maximum allowable difference.

4.3.2 Suspended solids and turbidity

Introduction

Increased suspended solids (SS) may cause a number effects on fish. These may be indirect or direct such as siltation of gills. Certain factors may increase the concentration of SS these include, afforestation, during the early stages of the forestry cycle and felling, and gravel extraction, which may cause the release of fine sediments.

Turbidity is an optical property of water whereby suspended and dissolved materials cause light to be scattered rather than transmitted in straight lines. An increase in turbidity has been shown to reduce light penetration in lakes and streams and is associated with a decrease in primary production of fish food organisms and of fish. Turbidity levels are often expressed as Nephelometric Turbidity Units (NTU) or Jackson Turbidity Units (JTU) which are approximately equal.

Standards applicable to the UK

There appear to be no current water quality criteria set for the protection of migratory fisheries, for either fresh or estuarine water. Standards that have been set or proposed for

resident fresh water fisheries are given in Sections 2.3.3 and 3.3.2, but these are designed to protect resident populations rather than migratory fish. Relevant data on the effects of SS and turbidity on migratory fish are given below.

Recent data

EIFAC (Alabaster and Lloyd 1982), reviewed the effect of SS on European fresh water fish, but did not propose standards for the passage of migratory fish. However, they cite four studies (discussed below) that indicate migration will still occur at high levels of SS, levels at which resident fisheries may be severely affected.

Atlantic salmon were netted in the River Severn estuary although parts of the estuary naturally contains SS up to several thousand mg/l at times, Gibson (1933). Smith and Saunders (1958) studied the movement of brook trout between fresh and salt water and found that turbidity appeared to have no effect on the movement of the fish. Ward (1938) found salmon runs occurred in Oregon streams, which had normal suspended solid concentrations of 137 - 395 mg/l. However, in contrast, Sumner and Smith (1939) found that king salmon avoided muddy water, and entered cleaner estuaries. The fish also chose clean areas rather than more turbid areas for spawning.

Lloyd *et al* (1987) reported two studies on the effect of high turbidity on migration. Meehan and Siniff (1962) suggested turbidity altered the daily pattern of downstream migration of salmon smolt by masking differences in the light between night and day. While Brett and Groot (1963) suggested that it may effect migration by interference of visual cues.

As previously mentioned in Section 2.3.2, Lloyd (1987) cited a number of studies that indicated avoidance by non-indigenous salmonids of waters with turbidities in the range 22 - 265 NTU.

Discussion

Studies suggest that even at high levels of SS, migration of salmonids will still occur. However, there are indications that particularly high levels may interfere with migration, either through interference with visual cues or due to the fish actively avoiding such waters when cleaner waters are available.

Because of the limited data, it is not possible to propose standards for SS or turbidity that will allow the migration of salmonids through estuaries or fresh water. It appears that SS or turbidity, unless very high, are unlikely to interfere with migration, with other factors likely to be limiting. It is therefore suggested that standards for suspended solids (or turbidity) be developed locally as necessary and based on local observations.

4.4 Biological standards

As previously stated in Section 2.4 while compliance with fisheries objectives could be assessed by monitoring for biological standards, a fish classification scheme for use throughout the NRA is currently under development (Mainstone and Wyatt 1991). Hence biological standards will not be proposed here.

4.5 Habitat

As mentioned in Section 2.5, it is not within the scope of this report to consider or propose criteria for habitats for migratory fisheries. However, habitat alteration may have the effect of drastically altering a fish population regardless of the chemical quality of the water. Thus it is a necessary parameter to consider when regarding the status of a fishery, with both high habitat and chemical quality needing protection in order to maintain an adequate fishery.

5. COMMERCIAL HARVESTING OF MARINE FISH FOR PUBLIC CONSUMPTION

5.1 Environmental quality objective

The EQO for commercial harvesting of marine fish for public consumption only applies to estuarine and marine waters. The objective is concerned with controlling the factors which may affect the quality of fish harvested for public consumption.

EQO

Explanatory Notes

Maintain water quality such that commercial marine fish quality shall be acceptable for human consumption as determined by the appropriate competent authorities (e.g. MAFF).

This objective relates only to suitability for human consumption. The general health of the fish themselves is protected under the General Ecosystems objective.

The setting of standards for the protection of marine harvested fish could be approached in two ways. Standards could be set that give maximum allowable concentrations of contaminant in fish flesh, or as a contaminant concentration in water that would prevent their accumulation in fish flesh. This Section has taken the approach of this second option by proposing 'Warning Levels' to prevent either maximum concentrations in foods or permissible intakes of contaminants, via fish flesh, being exceeded. Table 5.1 gives UK statutory or recommended standards for contaminant concentrations in fish flesh or permissible intakes, while Table 5.2 summarises the proposed warning levels along with the data used to calculate them.

The proposed warning levels are concentrations of contaminants in water, which, based on the available data, are the maximum concentrations that can be present in waters from which marine fish are harvested for human consumption without statutory or recommended standards being exceeded. It is acknowledged that for all the contaminants dealt with in this section, the data are limited and do not give an overall picture of the bioaccumulation potential of all fish species or biomagnification in food chains. However, the warning levels are based on highest reported Bioconcentration Factor (BCF) data and are designed to give an indication of contaminant concentrations in water above which harvested fish may result in statutory or recommended contaminant concentrations being exceeded. A weekly consumption figure (WCF) of 350 g fish per week has been used for calculation of warning levels. This is based on the figure for extreme consumption of fish, amongst fishing communities, reported by MAFF (1987), see Section 5.2. It is recommended that marine fish harvested from waters where the contaminant concentration exceeds the proposed warning level should be checked for accumulation of that contaminant.

Two different approaches have been taken to derive the proposed warning levels. The first is based on achieving standards or guidelines for the maximum permissible concentration of the contaminant in fish flesh. In this case the warning levels are calculated as follows:

$$WL_{(FS)} = \frac{CS}{BCF} \times 1000$$

where: $WL_{(FS)}$ = proposed warning level (in $\mu\text{g/l}$) for compliance with food/fresh standards or guideline

BCF = bioconcentration factor

FS = food/flesh standard or guideline (in μg).

The second approach is based on health-related acceptable/tolerable intakes of contaminants and the warning levels are calculated as below:

$$WL_{(m)} = \frac{PTWI}{BCF \times WCF}$$

where: $WL_{(m)}$ = proposed warning level (in $\mu\text{g/l}$) for weekly intake of contaminant to be below health-related tolerable intake guidelines.

BCF = bioconcentration factor

$PTWI$ = provisional tolerable weekly intake (in μg)

WCF = weekly consumption figure (0.35 kg).

The contaminants considered in this section are those that have had standards applicable to the UK set or recommended, or are contaminants previously mentioned in Smith *et al* (1991) for which numerical standards have not been set but their presence may give cause for concern. With regard to the setting of these warning levels, standards pertinent to the UK relating to maximum permissible levels of contaminant in foods or maximal permissible intakes have been reviewed, as has current available information on BCFs of marine fish. However, for some contaminants, data has been too limited to propose warning levels.

Table 5.1 Maximum allowable concentrations in fish flesh or permissible/acceptable human intake of contaminants applicable to the UK and fish consumption

Contaminant	MACF (mg/kg, wet weight)		Permissible/ acceptable intake (mg/kg, wet weight)	Reference
Mercury	0.3	(I)	-	3
	0.5		-	2
	-		0.005 (PTWI)	5
Cadmium	-		0.007 (PTWI)	5
Lead	2	(I)	-	4
	-		0.025 (PTWI)(child)	6
	-		0.04 (PTWI)(adult)	6
Arsenic	1(1)	(I)	-	1
			0.014 (PTWI)	5
Copper	20	(R)	-	2
Tin	250	(R)	-	2
Zinc	50	(R)	-	2
Permethrin	-		0.05 (ADI)	7

Notes: ADI Acceptable daily intake.
MACF Maximum allowable concentration in fish.
PTWI Provisional tolerable weekly intake.
(I) Mandatory concentration.
(R) Recommended concentration.
(1) Except where natural levels are higher.

References: 1. Arsenic Food Regulations (1959).
2. Food Standards Committee (in FAO 1983).
3. CEC (1982).
4. Lead Food Regulations (1979).
5. FAO/WHO (1984).
6. FAO/WHO (1986).
7. FAO/WHO (1984; cited in Zabel *et al* 1988a).

Table 5.2 Proposed warning levels of contaminant in water for harvesting of marine fish for public consumption (see text for discussion)

Contaminant	Food/flesh standard µg/g (1)	EQS µg/l	BCF (highest reported) (2)	EQS × BCF µg/g (in flesh) (3)	Proposed Warning Level µg/l (Dissolved) (for food standard)	PTWI µg (based on a 70 kg adult)	EQS µg/l	BCF (highest reported) (2)	EQS × BCF × WCF µg (4)	Proposed Warning Level µg/l (Dissolved) (for PTWI)
Mercury	0.3	0.3	4994	1.50	0.06	300	0.3	4994	524	0.2
Cadmium	-	2.5	1000	2.5	-	500	2.5	1000	875	1.4
Lead	2	25	12	0.3	nn	3000	25	12	105	nn
Arsenic	1	25	35	0.9	nn	1000	25	35	306	nn
Copper	20	5	nr	-	- (5)	-	5	nr	-	- (5)
Tin	250	10	nr	-	nn	-	10	nr	-	-
Zinc	50	40	128	5	nn	-	40	128	-	-
Permethrin	-	0.01	20 000	0.2	-	24 500	0.01	20 000	70	nn (2)
TBT	-	0.002	na (2)	-	-	-	0.002	na (2)	-	nn (2)
TPT	-	0.008	na (2)	-	-	-	0.008	na (2)	-	nn (2)
PCB	-	-	-	-	-	-	-	-	-	0.00079 (6)
Total phenol	-	-	-	-	-	-	-	-	-	1.0 (7)

Notes (1) Mandatory or recommended, see Table 5.1.

(2) See appropriate section in text for explanation.

(3) This column shows the maximum likely concentration of contaminant in fish flesh if the EQS is just achieved. When greater than the food/flesh standard a warning level is necessary.

(4) This column shows the maximum likely weekly intake of contaminant if the EQS is just achieved. When greater than the PTWI a warning level is necessary.

(5) - in this column indicates insufficient data to propose a warning level.

(6) Based on US EPA criteria, see Section 5.7.

(7) Based on CCREM guideline, see Section 5.9.

na Not applicable.

nn Not necessary (i.e. would be greater than EQS for protection of salt water life).

nr Not reported.

WCF Weekly consumption figure (350 g).

TBT Tributyltin.

TPT Triphenyltin.

5.2 Mercury

Standards have been set by the CEC and WHO for the maximum concentration of mercury in fish. The CEC (1982) sets a specific value for the concentrations of mercury which states that the concentration of mercury in "a representative sample of fish flesh chosen as an indicator" must not exceed 0.3 mg/kg wet flesh, however, it is not clear on what basis this standard was set. A similar standard of 0.5 mg/kg wet weight for the acceptable concentration of mercury in fish has been recommended by WHO (FAO 1983). Data provided by the FAO (1983) and SPCC (1990) on maximum permissible levels of mercury in fish flesh showed only one standard to be more stringent than those proposed here, this was a maximum level of 0.1 mg/kg (fresh weight) for sardines in Venezuela.

MAFF (1987) reported the national average weekly consumption of fish in the UK to be 140 g, however, for certain high consumption groups, this increased to an average of 350 g per week. For the purpose of this report calculations involving contaminant intake will use 350 g as an estimate of weekly consumption. The FAO/WHO provisional tolerable weekly intake (PTWI) for mercury is 0.005 mg/kg body weight, equivalent to 0.3 mg based on a 70 kg adult (FAO/WHO 1984). This allows for a weekly consumption of 1000 g of fish reaching the 0.3 mg/kg standard set by the CEC. Thus it can be concluded that this standard would be adequate in protecting human consumers of fish.

Maximal acceptable concentrations of mercury in water for the protection of human consumers of fish have been proposed by both the US EPA (cited by SPCC 1990) and the CCREM (1987). The US EPA have proposed a concentration of 0.15 µg Hg/l for the protection of consumers of marine fish, while the CCREM have recommended a slightly lower concentration of 0.1 µg/l total mercury for the protection of consumers of fresh water fish. No such standard has been set in the UK, but the Surface Waters (Dangerous Substances) (Classification) Regulations (HMSO 1989) sets a maximum standard of 0.3 µg/l dissolved Hg in estuaries and coastal waters.

At the current maximum standard of 0.3 µg Hg/l for estuarine and coastal waters (HMSO 1989) the CEC mandatory standard of 0.3 mg/kg wet flesh, would be exceeded if a BCF of only 1000 was reached, Jones *et al* (1988) reported BCFs exceeded this in a number of aquatic organisms and thus it is probable that a number of fish in estuarine and marine waters would have BCFs in excess of this. Data on BCF for mercury in marine fish are limited, both Jones *et al* (1988) and WHO (1989a) only reported BCFs for fresh water fish, the highest reported BCF for mercury being 4994 for fathead minnow (CCREM 1987).

Calculating water concentrations of mercury at which the PTWI and flesh standards would be reached indicates that, using the highest reported BCF of 4994 and an average weekly consumption of 350 g fish, a water concentration of 0.2 µg/l would result in the PTWI of 0.3 mg (for a 70 kg adult) (FAO 1984) just being exceeded. While, again based on a BCF of 4994, a water concentration of 0.06 µg/l would result in the CEC mandatory standard of 0.3 mg/kg (wet weight) for fish flesh being reached.

Due to the absence of information on BCFs for mercury in marine fish, it is difficult to assess if they would be higher or lower than those reported for fresh water fish and hence in order to provide some safety margin the highest BCF reported, that for the freshwater fish

fathead minnow, has been used. It appears that the current maximum water concentration for estuaries and marine waters (HMSO 1989) is too high for adequate protection of human consumers of estuarine and marine fish. It is therefore proposed that 0.06 µg/l dissolved Hg be used as a warning concentration in order to prevent the CEC mandatory mercury standard for fish being exceeded, this should also prevent the PTWI being exceeded.

In addition, as previously stated in Smith *et al* (1991) the FAO/WHO PTWI for mercury of 0.3 mg (based on a 70 kg adult) should not include more than 0.2 mg as methylmercury (MeHg). Therefore it is proposed that a proviso should be added to the mandatory EC standard to limit the methylmercury concentration to 0.2 mg MeHg-Hg/kg.

5.3 Cadmium

There are no statutory maximum concentrations for cadmium in food set for the UK, but a PTWI of approximately 7 µg/kg body weight, equivalent to a weekly intake of approximately 500 µg for a 70 kg adult, has been recommended by WHO (FAO/WHO 1984).

In addition, standards for cadmium in water have been set by the CEC (1983), these are maximum concentrations of 5 µg/l in estuarine waters and 2.5 µg/l in marine waters (dissolved annual averages). However, the Surface Waters (Dangerous Substances) (Classification) Regulations (HMSO 1989) have set a stricter maximum standard concentration of 2.5 µg/l (dissolved annual average) for cadmium in estuarine water, the same as for marine waters.

BCFs in excess of 1000 have been reported by Jones *et al* (1988) for marine fish, additional data since Jones *et al* (1988) on BCF of marine fish for cadmium are not available. Based on these data, in waters where the concentration of cadmium was 2.5 µg/l (the maximum allowable concentration according to the Dangerous Substances Regulations, HMSO 1989) and bioaccumulation factors in fish of 1000, a concentration of 2.5 mg/kg cadmium would accumulate in fish. At this concentration, a weekly intake of 200 g fish flesh would result in the PTWI being reached and thus the weekly consumption of 350 g of fish per adult would exceed the PTWI.

Hence it appears that the current allowable maximum concentration of cadmium in estuarine and marine waters, 2.5 µg/l (HMSO 1989), is not adequate to protect human consumers of marine and estuarine fish. Therefore, in order to prevent the PTWI of cadmium being exceeded, a warning concentration of 1.4 µg/l dissolved cadmium is proposed. At this concentration fish with a BCF of 1000 would accumulate cadmium below a level that would cause a consumption of 350 g fish per week to exceed the PTWI of 500 µg. It is proposed that fish harvested from waters in excess of this concentration should be checked to ensure that accumulated concentrations of cadmium do not exceed the PTWI.

5.4 Arsenic

The Arsenic in Food Regulations 1959, lay down a statutory limit of 1 mg/kg for the concentration of arsenic in food, this applies to fish, except where natural levels are higher. Only one standard from other countries appears to be lower (SPCC 1990). This is 0.1 mg/kg (fresh weight) maximal permissible levels in tuna and sardines set in Venezuela where these fish are a major part of the diet.

A maximum acceptable daily intake of 0.002 mg/kg body weight, has been recommended by FAO/WHO (1984) equivalent to a weekly intake for a 70 kg adult of approximately 1 mg. In addition, WHO (1981) consider an intake of 0.4 mg/day of inorganic arsenic over a lifetime relates to a 5% prevalence of arsenic induced skin cancer.

Friberg (1988) stated that 10% of the arsenic in fish is present in the inorganic form, the remainder being the relatively non-toxic arsenobetaine. Thus a weekly intake of 350 g of fish reaching the UK statutory limit of 1 mg/kg would result in an inorganic arsenic intake of 0.035 mg/week. This is below the WHO standard (1981) of 0.4 mg/day inorganic arsenic, relating to a 5% prevalence of arsenic induced skin cancer.

Standards relating to the concentration of arsenic in water for the protection of human consumers of marine fish are limited. Only the US EPA, (cited in SPCC 1990) have proposed such a standard, a concentration of 17.5 ng As/l, a BCF of >57 000 would be necessary before the UK statutory standard was exceeded. No such standard has been set in the UK, however, an EQS of an annual average dissolved concentration of 25 µg/l arsenic for the protection of salt water life has been set (DoE 1989). In waters reaching this EQS, a BCF of 400 would be necessary for the UK statutory limit of 1 mg/kg arsenic in food to be exceeded. However, there are no available data on BCF factors for marine fish to assess if this standard would adequately protect human consumers. Mance *et al* (1984a) reported BCF for fresh water species of fish to be in the range 1 to 35, however the lack of data make it difficult to assess the accumulation potential of marine fish.

It is therefore tentatively proposed in the light of the low bioaccumulation of arsenic by fresh water fish, that the current EQS of 25 µg/l dissolved arsenic for the protection of salt water life should also be adequate for the protection of human consumers of estuarine and marine fish and thus it does not appear necessary to propose a warning level above which harvested fish should be monitored.

5.5 Lead

The Lead in Food Regulations (1979) lay down statutory limits on the allowable concentration of lead in food, for fish, the concentration of lead must not exceed 2 mg/kg wet weight. In addition, to the statutory level, the FAO/WHO (1986) have recommended PTWIs of 25 µg/kg for children and 3 mg for adults (based on a 70 kg adult). Food standards for lead reported from other countries (SPCC 1990) showed some to be marginally more stringent. Standards ranged from 1.5 mg/kg in certain states of Australia (2 mg/kg in other states) down to 0.5 mg/kg in the Netherlands and Philippines.

Based on the PTWI for a 10 kg child (250 µg), an acceptable weekly intake of fish with a lead concentration at the statutory level of 2 mg/kg (ww) would be 125 g fish/week, while for adults, the PTWI permits a intake of 1500 g of fish just meeting the statutory limit. Thus according to the average consumption of 350 g fish, reported for certain high intake groups (MAFF 1987), for adults, the intake of lead from fish would not appear to be a problem, while for children consuming regularly higher amounts of fish containing these levels, there may be a risk.

No standards for lead concentration in water for the protection of human consumers of marine fish have been set in the UK, or in other countries. The DoE (1989) has adopted an EQS for lead as a dissolved annual average of 25 µg/l (proposed by Brown *et al* 1984) for the protection of salt water life. Brown *et al* (1984) found marine fish did not appear to bioaccumulate lead in their muscles, the highest BCF reported was 12 for tri-ethyl lead in the muscle of dab (*Limanda limanda*). This is in contrast to fresh water fish for which Brown *et al* (1984) reported BCFs in the range 100 to 1000. The authors stated that based on the available data, they could not be certain, but they did suggest, that the EQS of 25 µg/l dissolved annual average would ensure that lead in the muscle of marine and estuarine fish would not exceed the 2 mg/kg standard.

Additional data since Brown *et al* (1984) on bioaccumulation of lead in estuarine and marine fish are limited. There are no additional data on BCFs in fish, but a study by Jaffar and Ashraf (1988) appears to confirm that bioaccumulation of lead in the muscles of fish is unlikely to be a problem. Jaffar and Ashraf (1988) investigated the uptake of lead in different tissues of twelve species of fish from the coastal waters of Pakistan. While the distribution of the metals in different species was found to be species-specific, in all cases, levels were always lowest in muscles and highest in kidney or liver. In addition, WHO (1989b) concluded that lead uptake in fish reached equilibrium only after a number of weeks of exposure, and that it is accumulated mostly in the gill, liver, kidney and bone.

In order for the statutory limit of 2 mg/kg lead in fish flesh to be exceeded in waters reaching the EQS of 25 µg/l dissolved lead, it would be necessary for BCF in muscle to reach 80. BCFs as high as this have not been reported for estuarine or marine fish, but they have for fresh water fish. It is therefore proposed that while it would appear that the current EQS of 25 µg/l for the protection of salt water life will enable the statutory standards of 2 mg/l Pb in fish to be met and therefore adequately protect human consumers, fish harvested from waters reaching 25 µg/l should be monitored for lead accumulation to ensure that this is so. This is in preference to setting a warning level of 2 µg/l based on the BCF of 1000 reported for fresh water fish since the available data shows that lead will not be accumulated in the muscle of estuarine or marine fish.

5.6 List II substances

The following List II substances have been reviewed in WRc Technical Reports and their estimated intake from fish is not considered to present a significant risk to human health:

- boron, Mance *et al* (1988a);
- chromium, Mance *et al* (1984b);
- copper, Mance *et al* (1984c);
- inorganic tin, Mance *et al* (1988b);
- iron, Mance and Campbell (1988);
- nickel, Mance and Yates (1984a);

- vanadium, Mance *et al* (1988d);
- zinc, Mance and Yates (1984b).

There are no statutory limits for the protection of fish consumers in force in the UK, thus while EQSs for the protection of salt water fish have been proposed and/or adopted for these substances, it is not of relevance to review the bioaccumulation potential in fish with regard to assessing if the EQS will prevent exceedance of standards in fish flesh.

However, maximum levels of copper, zinc and tin in food have been recommended by the Food Standards Committee (reported by FAO 1983) and these are further outlined in Sections 5.6.1 - 5.6.3.

In addition, the environmental effects of mothproofing chemicals (Zabel *et al* 1988a) and organotins (Zabel *et al* 1988b) in water have been reviewed.

Considering the WHO/FAO acceptable daily intake (ADI) (FAO/WHO 1984 cited in Zabel *et al* 1988a) for permethrin of 50 µg/kg/day (equivalent to 3.5 mg/day or 24.5 mg/week for a 70 kg adult), the highest reported bioaccumulation factor for rainbow trout, 387 and a daily consumption of 100 g/day fish flesh, Zabel *et al* (1988a) concluded that fish harvested from waters complying with the EQS for the protection of salt water life, 10 ng/l total permethrin (95 percentile), would be unlikely to be harmful to human health.

Information on BCFs for salt water species is limited to sheepshead minnow with Zabel *et al* (1988a) reporting a BCF of 620. Based on an average weekly intake of 350 g fish flesh (MAFF 1987), fish with such a BCF harvested from waters reaching the EQS of 10 ng/l (total active ingredient), would result in a weekly intake of 2.17 mg permethrin. With the exception of one study on fathead minnows (Zabel *et al* 1988a) where a BCF of 20 000 was reported, BCFs for fresh water fish were such that an average weekly consumption of fish flesh harvested from waters of 10 ng/l permethrin, would still be within the ADI. Thus, as concluded in Zabel *et al* (1988a) the EQS of 10 ng/l (total active ingredient) should be adequate to protect human consumers, and no warning level of permethrin for the harvesting of estuarine and marine fish is proposed.

ADIs for the other mothproofing chemicals (PCSDs, cyfluthrin, flucofenuron, sulcofenuron and HHP) have not been agreed so such calculations are not possible.

In addition, Zabel *et al* (1988b) concluded that the EQSs for tributyl- and triphenyltin (2 and 8 ng/l total, maximum allowable concentration) for the protection of salt water life would be adequate for the protection of human consumers of fish. This was on the basis that fish and shellfish would be affected at lower levels than were likely to be harmful to humans.

5.6.1 Copper

Copper has been considered in a WRc Technical Report (Mance *et al* 1984c) which considered that the intake of copper from fish consumption would not present a significant risk to human health. While there are no legal limits in force in the UK for the protection of fish consumers, the Food Standards Committee have recommended a maximum level of

20 mg/kg copper in food, which is applicable to fish (FAO 1983). Standards for maximum levels in fish from other countries are in the range 10 - 30 mg/kg (SPCC 1990).

Mance *et al* (1984c) proposed an EQS for the protection of salt water fish and shellfish of 5 µg/l copper as dissolved annual average concentration, this has since been adopted by the DoE (1989) for the protection of salt water life.

However, currently, due to lack of available data, it is difficult to assess if marine fish will accumulate copper from waters complying with the EQS in excess of the recommended food standard. No data on the BCF for fish were reported in Mance *et al* (1984c), however, WRc is currently reviewing the EQS for copper and further bioaccumulation data may be available when this is released. Thus at present it is not possible to assess if the guideline would be exceeded by fish harvested from waters just achieving the EQS.

5.6.2 Tin

Tin has been considered in a WRc Technical Report (Mance *et al* 1988b) and the intake from fish consumption was considered not to present a significant risk to human health. While there are no legal limits in force in the UK for the specific protection of fish consumers, the Food Standards Committee have recommended a maximum level of 250 mg/kg tin in food (FAO 1983) this is applicable to fish intended for human consumption. Standards for maximum levels in fish intended for human consumption from other countries are in the range 50 - 250 mg/kg (SPCC 1990).

Mance *et al* (1988b) proposed an EQS of 10 µg Sn/l as a dissolved annual average for the protection of salt water fish and shellfish, but this has not yet been adopted by the DoE (1989). It is not possible to consider if the proposed EQS is low enough to prevent accumulation of tin in fish to levels exceeding the recommended food guideline, since Mance *et al* (1988b) reported no data on BCF in marine fish, and none appear to have been reported since. However, as previously mentioned, the estimated intake of tin from fish is not considered a hazard, and thus it can only be suggested that human consumers of fish harvested from waters complying with EQS of 10 µg/l dissolved Sn will be adequately protected.

5.6.3 Zinc

Zinc has been considered in a WRc Technical Report, Mance and Yates (1984b), it was considered that intake from marine fish consumption did not present a significant risk to human health. No legal limits are in force in the UK specifically for the protection of fish consumers, but the Food Standards Committee have recommended a maximum level of 50 mg/kg in food and this is applicable for fish intended for human consumption (FAO 1983).

Mance and Yates (1984b) proposed an EQS for the protection of salt water fish and shellfish of 40 µg/l as an annual average dissolved concentration. This standard has since been adopted by the DoE (1989) for the protection of salt water life.

Mance and Yates (1984b) found no laboratory data on bioaccumulation factors for marine fish, and field data indicated that zinc may either accumulate or alternatively be regulated.

Data not reported in Mance and Yates (1984b) from Sauer and Watabe (1984) found that the fish mummichog (*Fundulus heteroclitus*) when exposed to 0.21 mg/l (dissolved) Zn for eight weeks, accumulated a whole body concentration of 198 µg/g Zn (dry weight; dw), this corresponds to a BCF (dw) of 943. However, a much higher BCF of 2560 can be calculated for plaice (*Pleuronectes platessa*) from the data of Milner (1982), who reported, after 139 days exposure to 0.05 mg/l (dissolved) Zn, a tissue concentration of 128 µg/g Zn (dry weight).

Considering the dry weight BCF for plaice and applying this to waters reaching the EQS of 40 µg/l, suggests that zinc may accumulate to a concentration of around 100 mg/kg, twice the recommended limit. However, assuming that ash weight is approximately 5% of fresh weight, this gives a BCF of 128 (wet weight; ww) and an equivalent concentration of 5 mg/kg in flesh (ww), ten times below the recommended level of 50 mg/kg.

It is therefore proposed in the light of the limited data, that the EQS of 40 µg/l dissolved zinc for the protection of salt water life will adequately protect human consumers of fish. Fish harvested from waters reaching this limit should not accumulate zinc to levels that will exceed that recommended for food, thus a warning level is not proposed.

5.7 Organochlorines

Smith *et al* (1991) concluded that with the exception of polychlorinated biphenyls (PCBs), organochlorines, while highly persistent and with a high bioaccumulation potential, are well regulated in UK estuarine and marine waters and thus would not present a threat to human consumers of marine fish.

No standards for allowable concentrations of PCBs in food are in force or recommended in the UK and none appear to be available for other countries.

The US EPA (1986) concluded that since PCBs are potential human carcinogens the ambient concentration of PCBs in water should be zero, however, recognising the impracticabilities of this standard, they suggested a level of 0.79 ng/l would reduce the risk of cancer to people consuming aquatic organisms from such water. It is therefore proposed that this concentration, 0.79 ng/l PCB be used as a warning level, and that fish harvested from waters exceeding this concentration be checked for bioaccumulation.

5.8 Radionuclides

Ackefors *et al* (1990) have recently reviewed the accumulation of radionuclides by marine fish. With the exception of marine fish harvested from areas near a point source of radionuclides, there does not appear to be a problem of accumulation of these nuclides in fish. However, warning levels for radionuclides for the protection of human consumers of marine fish are not proposed due to the range of nuclides that may be present from such sources, and the limited data on the bioaccumulation potential of various fish.

Instead warning levels should be based on local observations where there are known discharges of radionuclides in effluents, or high background levels. Derivation of the levels should include consideration of the radionuclide present, the fish harvested from that area, levels accumulated in harvested fish and standards proposed by the International Commission on Radiological Protection (ICRP) for the maximum acceptable exposure via a particular pathway. The ICRP has recommended an annual dose limit of 1 mSv (millisievert) for humans. This dose is the sum of the dose received from external radiation in one year and that received from intakes of radionuclide in the same year (NRPB 1986).

5.9 Tainting substances

Materials should not be present in concentrations that individually or in combination produce undesirable flavours or odour in the edible portion of fish.

Both SPCC (1990) and Persson (1984) have reported ranges of estimated threshold concentrations (ETCs), (the lowest concentration in the waters that impairs the flavour of the fish), of individual and mixtures of substances in water found to cause tainting in fish flesh. However, the data from Persson (1984) showed large variations due to differences in fish species, exposure conditions, sensory methods and the quality of chemicals used. ECETOC (1987) reported that there has been little critical assessment of the test methods used for the derivation of ETCs and that there was a need for more research to firmly establish ETCs for potential tainting substances. The European Chemical Industry Ecology and Toxicology Centre (ECETOC) has developed and tested a procedure for the evaluation of the tainting potential of substances to seafood. Such a procedure would be useful in the further evaluation of ETCs.

There are no specific standards in the UK relating to the concentration of substances that impair or taint the flavour of fish flesh. The CEC (1978), in the Fresh Water Fish Directive, only state that phenolic compounds should not be present at concentrations that adversely affect the flavour of the fish and petroleum hydrocarbons should not be present at concentrations that impart a detectable hydrocarbon taste to the fish, numerical values were not given.

A few guideline standards for the concentration of phenols in water in order to prevent tainting of fish flesh have been proposed. The Ontario Ministry of the Environment (McKee *et al* 1984, cited in the CCREM (1987)) have recommended that in order to prevent the tainting of fish flesh, the concentration of monochlorophenols and dichlorophenols should not exceed 7 and 0.2 µg/l respectively. These criteria are half of the tainting thresholds found for these compounds. In addition, the CCREM (1987) state that in order to prevent the tainting of fish flesh, the concentration of total phenols should not exceed 1 µg/l, however, the standard was proposed for fresh water organisms.

No standards have been proposed for petroleum hydrocarbons. SPCC (1990) reported ETCs of various petroleum products to be in the range 0.005 - 1 mg/l, while Persson (1984) reported the concentration of petroleum products that caused 'off flavours' in fish to be in the range 0.1 - 833 mg/l.

A wide variety of substances have been found to cause tainting of fish flesh, but standards proposed in other countries are limited, and no numerical standards are available for the UK. In the absence of data it is proposed that the CCREM standard (1987) of 1 µg/l for total phenols be applied as a warning level in waters used for the harvesting of marine fish in order to reduce the risk of tainting fish flesh. Warning levels for other chemicals known to cause tainting in fish cannot be proposed because of the lack of evaluation of methods and the need for further research in this area.

6. COMMERCIAL HARVESTING OF SHELLFISH FOR PUBLIC CONSUMPTION

6.1 Environmental quality objective

The EQO for commercial harvesting of shellfish for public consumption only applies to estuarine and marine waters. The objective is concerned with controlling the factors which may affect the quality of shellfish harvested for public consumption.

EQO

Explanatory notes

Maintain water quality such that commercial shellfish quality shall be acceptable for consumption as determined by the appropriate authorities (e.g. MAFF).

This objective relates only to suitability of shellfish for human consumption, the general health of the shellfish themselves is protected under the General Ecosystem objective.

As with marine fish harvested for public consumption, the setting of standards for the protection of this objective could be approached in two ways. Standards could be set that give maximum allowable concentrations of contaminant in shellfish flesh, or as a contaminant concentration in water that would prevent their accumulation in shellfish flesh. This section has taken the approach of this second option by proposing warning levels to prevent either maximum concentrations in foods or permissible intakes of contaminants, via fish flesh, being exceeded. Table 6.1 gives UK statutory or recommended standards for contaminant concentrations in shellfish flesh or permissible intakes, while Table 6.2 summarises the proposed warning levels along with the data used to calculate them.

As previously explained in Section 5.1 the proposed warning levels are concentrations of contaminants in water, which, based on the available data, are the maximum concentrations that can be present in waters from which shellfish are harvested for human consumption without statutory or recommended standards being exceeded. As before, it is acknowledged that for all the contaminants dealt with in this section, the data are limited and do not give an overall picture of the bioaccumulation potential of all shellfish species or biomagnification in food chains. However, the warning levels are based on highest reported BCF data and are designed to give an indication of contaminant concentrations in water above which harvested shellfish may result in statutory or recommended contaminant concentrations being exceeded. In the absence of any data on amounts of shellfish consumed in the UK, the figure for fish consumption, 350 g/week, has been used to calculate warning levels. It is recommended, that shellfish harvested from waters where the contaminant concentration exceeds the proposed warning level should be checked for accumulation of that contaminant.

The contaminants considered in this section are those that have had standards applicable to the UK set or recommended, or are contaminants previously mentioned in Smith *et al* (1991) for which numerical standards have not been set but their presence may give cause for

concern. With regard to the setting of these warning levels, standards pertinent to the UK relating to maximum permissible levels of contaminant in foods or maximal permissible intakes have been reviewed, as has current available information on BCFs of marine fish. However, for some contaminants, data have been too limited to propose warning levels.

Table 6.1 Maximum allowable concentrations in shellfish flesh or permissible/acceptable human intake concentrations of contaminants applicable to the UK and shellfish consumption

Contaminant	MACF(mg/kg wet weight)	Permissible/ acceptable intake (mg/kg, wet weight)	Reference
Mercury	0.5	- 0.005 (PTWI)	2 4
Cadmium	-	0.007 (PTWI)	4
Lead	10 (I)	-	3
	-	0.025 (PTWI) (child)	5
	-	0.04 (PTWI) (adult)	5
Arsenic	1(1) (I)	-	1
	-	0.014 (PTWI)	4
Copper	20 (R)	-	2
Tin	250 (R)	-	2
Zinc	50 (R)	-	2
Bacteria (per 100 g mollusc flesh)(for direct consumption)	300 fc	-	6
(if to be treated before placed on market)	6000 fc (2)	-	6
	60 000 fc (2)	-	6
Enteric Viruses	0	-	6

Notes: fc Faecal coliforms.
MACF Maximum allowable concentration in fish.
PTWI Provisional tolerable weekly intake.
(I) Mandatory concentration.
(R) Recommended concentration.
(1) Except where natural levels are higher.
(2) See Section 6.10 for definition of treatment requirements.

References: 1. Arsenic Food Regulations (1959).
2. FAO (1983).
3. Lead Food Regulations (1979).
4. FAO/WHO (1984).
5. FAO/WHO (1986).
6. CEC (1991).

Table 6.2 Proposed warning levels of contaminants in water for harvesting of shellfish for public consumption

Contaminant	Food/flesh standard µg/g (1)	EQS µg/l	BCF (highest reported) (2)	EQS × BCF µg/g (in flesh) (3)	Proposed warning level µg/l (dissolved) (for food standard)	PTWI µg (based on a 70 kg adult)	EQS µg/l	BCF (highest reported) (2)	EQS × BCF × WCF (µg) (4)	Proposed warning level µg/l (dissolved) (for PTWI)
Mercury	0.5	0.3	nr	- (2)	0.15 (5)	-	0.3	nr	-	-
Cadmium	-	2.5	1000	2.5	-	500	2.5	1000	875	1.4
Arsenic	1	25	200	5	5	1000	25	200	1750	14
Lead	10	25	1000	25	10	3000	25	1000	8750	8
Copper	20	5	194 000	970	0.25	-	5	194 000	-	-
Tin	250	10	nr	-	nn	-	10	nr	-	-
Zinc	50	40	30 950	1238	1.5	-	40	30 950	-	-
PCB	-	-	-	-	-	-	-	-	-	0.00079 (2)
Algal Toxins	80 (6)	-	-	-	-	-	-	-	-	- (2)

Notes (1) Mandatory or recommended, see Table 5.1.

(2) See appropriate section in text for explanation.

(3) This column shows that maximum likely concentration of contaminant in fish flesh if the EQS is just achieved. When greater than the food/flesh standard a warning level is necessary.

(4) This column shows the maximum likely weekly intake of contaminant if the EQS is just achieved. When greater than the PTWI a warning level is necessary.

(5) Total concentration.

(6) Standard as µg/100 g mollusc flesh.

nn Not necessary (i.e. would greater than EQS for protection of salt water life).

nr Not reported.

WCF Weekly consumption figure (350 g).

6.2 Mercury

The only food standard for mercury in the UK that applies to shellfish is the WHO guideline of 0.5 mg/kg. The standards set by CEC (1982) of 0.3 mg/kg applies specifically to fish flesh and therefore does not apply here.

No data on BCFs of mercury by shellfish appear to be available. Ackefors *et al* (1990) reported concentrations of mercury in bivalves could be as high as 10 mg/kg, but such concentrations were usually close to major point sources, with less than 50% consisting of methylmercury, no data on the concentration of mercury in the surrounding water column were available.

As mentioned in Section 5.2 a water concentration of 0.15 µg/l total mercury was proposed by the US EPA (cited in SPCC 1990) for the protection of human consumers of marine fish, crustaceans and molluscs. No such standard has been set in the UK, but the Surface Waters (Dangerous Substances) (Classification) Regulations (HMSO 1989) sets a maximum standard of 0.3 µg/l dissolved mercury (annual average) in estuaries and coastal waters. Because of the lack of BCF data for shellfish it is not possible to assess if shellfish harvested from waters of this concentration would conform to the food standard. It is therefore proposed, in the absence of other data that 0.15 µg/l total mercury be used as a warning level for the protection of human consumers of shellfish since this standard was originally proposed in the US for the protection of human consumers. In waters where this level is exceeded it is suggested that shellfish be examined for mercury accumulation.

Because the ratio of the FAO/WHO PTWI for mercury is 0.2:0.3, methylmercury:total mercury, it is suggested that based on the WHO guideline of 0.5 mg/kg, the concentration of methylmercury in shellfish reaching this limit should not exceed 0.33 mg/kg (i.e. $0.5 \times 2/3$).

6.3 Cadmium

As mentioned in Section 5.3, no statutory standards for cadmium concentrations in food have been set in the UK. However, the WHO (1989c) has established a PTWI of 7 µg/kg body weight, equivalent to a weekly intake of approximately 500 µg for a 70 kg adult.

No standards for cadmium concentrations in water for the protection of human consumers of shellfish have been set. The CEC (1983) have set maximum dissolved concentrations of 5 µg/l and 2.5 µg/l cadmium (annual averages) in estuaries and marine waters respectively. However, DoE have set a stricter standard of 2.5 µg/l for estuarine waters, the same as for marine waters (HMSO 1989).

As for marine fish, Jones *et al* (1988) have reported BCFs for marine shellfish in excess of 1000 (not reported if dry or wet weight). No further information since Jones *et al* (1988) on BCFs for cadmium in shellfish are available. Thus in waters with a concentration of 2.5 µg/l dissolved cadmium shellfish with a BCF of 1000 (assumed ww) would accumulate cadmium to a concentration of 2.5 mg/kg, allowing for a weekly consumption of 200 g of shellfish flesh before the WHO PTWI was exceeded. Unlike marine fish (Section 5.3), there is no available information on average weekly consumption of shellfish. In the absence of such information, it is proposed that the average weekly consumption of marine fish, that of 350 g

for certain groups, (MAFF 1987), be used as an average weekly consumption for shellfish, it is acknowledged that this may be an overestimation.

Hence it appears that the current allowable maximum concentration of dissolved cadmium in estuarine and marine waters, 2.5 µg/l (HMSO 1989), may not be adequate to protect human consumers of shellfish. In order to prevent the PTWI of cadmium being exceeded for certain groups of shellfish consumers, a warning concentration of 1.4 µg/l dissolved cadmium is proposed. At this concentration shellfish with a BCF of 1000 would probably accumulate cadmium to a level that would allow a consumption of 350 g shellfish per week to be below the PTWI of 500 µg for a 70 kg adult. It is proposed that shellfish harvested from waters in excess of this concentration be checked to ensure that cadmium has not accumulated to concentrations that may result in the PTWI being exceeded.

6.4 Arsenic

As previously stated in Section 5.4 a statutory standard of 1 mg/kg arsenic in foods has been set in The Arsenic in Food Regulations (HMSO 1959), this standard is applicable to shellfish. In addition, the WHO/FAO have estimated a PTWI for arsenic of 0.015 mg/kg body weight, equivalent to a weekly intake for a 70 kg adult of approximately 1 mg.

Only the US EPA (cited in SPCC 1990), who proposed a standard of 17.5 ng As/l, have given a specific standard for water concentrations of arsenic for the protection of human consumers of shellfish. At this concentration a BCF in excess of 57 000 would be needed before the statutory food standard was exceeded. No such water concentration standards have been proposed in the UK, however, Mance *et al* (1984a) proposed an EQS of 25 µg/l arsenic, (annual average dissolved concentration) for the protection of salt water life, this has since been adopted (DoE 1989).

No BCFs were reported by Mance *et al* (1984a), since then BCFs for molluscs of 3000 - 4000 (dry weight) have been reported by Rosemarin *et al* (1985) and Langston (1984). Shellfish with such BCFs harvested from areas corresponding to arsenic concentrations of 25 µg/l would exceed the statutory standard of 1 mg/kg. Using a BCF of 200 to estimated body concentrations (5% of 4000 to convert to wet weight, assuming that ash weight is approximately 5% of fresh weight), the statutory standard of 1 mg/kg would be reached at a water concentration of 5 µg/l.

Also at the current EQS of 25 µg/l, an estimated intake of 350 g shellfish per week (see Section 6.3) of shellfish with wet weight BCFs of 200 would result in the PTWI being exceeded. A water concentration of approximately 14 µg/l would be needed at this shellfish consumption and BCF to prevent the PTWI being exceeded.

Therefore in order to ensure that the statutory food standard of 1 mg/kg is not exceeded the stricter warning level of 5 µg/l dissolved arsenic is proposed. Shellfish harvested from waters exceeding this limit should be investigated for arsenic accumulation.

6.5 Lead

The Lead in Food Regulations (HMSO 1979) lay down a statutory standard of 10 mg/kg for shellfish foods. The FAO/WHO PTWI for adults (based on a 70 kg adult) is 3.0 mg, while that for children is 25 µg/kg. Based on these PTWIs, adults consuming a weekly average of 350 g shellfish at the statutory concentration would exceed the PTWI, while a 10 kg child would only need to consume 25 g shellfish at the statutory level before reaching the PTWI. Thus at least for children and for adults consuming high amounts of shellfish the statutory limit may not be strict enough to prevent the PTWI being exceeded.

Standards for other countries for lead in shellfish (FAO 1983, SPCC 1990) are more stringent than for the UK. The standards, which either encompassed all shellfish or just specific ones e.g. mussels, were in the range 0.5 mg/kg to 2 mg/kg.

Brown *et al* (1984) reported that bioaccumulation factors for lead in molluscs were typically about 1000 (not reported if wet or dry weight). More recent data have reiterated this bioaccumulation of lead in shellfish. Amiard-Triquet *et al* (1988) reported a maximum bioaccumulation factor (based on dry weight) of 3259 after four weeks for the Pacific oyster, *Crassostrea gigas*, following direct exposure to lead in water. In addition, Lakhmanan and Nambisan (1989) reported a bioaccumulation factor of 459 (wet weight) in the mussel, *Perna viridis*, when exposed to lead nitrate for six days.

An EQS of 25 µg/l (annual average dissolved concentration of lead) has been adopted by the DoE for the protection of salt water organisms (DoE 1989). Shellfish with a BCF of 1000 in waters of this concentration would probably accumulate concentrations of lead in excess of the statutory food standard. Therefore it is proposed that a water concentration of 10 µg/l dissolved lead be used as a warning level in order to prevent lead accumulating in shellfish in excess of the mandatory food standard. This concentration should result in shellfish with BCF of 1000 just reaching the statutory limit of 10 mg/kg. Shellfish harvested from waters above this concentration of lead should be checked for bioaccumulation.

Considering a BCF of 1000, more stringent water concentrations of approximately 8 µg/l and 0.7 µg/l lead would be needed in order to prevent exceedance of the PTWI for a 70 kg adult and a 10 kg child. However, because the estimated intake of shellfish is unsure and has been based on weekly marine fish intake, it is proposed that initially a warning level of 10 µg/l dissolved lead be used in order to prevent exceedance of the mandatory food standard.

6.6 List II substances

As previously stated in Smith *et al* (1991), the following List II substances have been reviewed in WRc Technical Reports and their estimated intake from shellfish is not considered to present a significant risk to human health.

- Boron, Mance *et al* (1988a);
- Chromium, Mance *et al* (1984b);
- Copper, Mance *et al* (1984c);

- Inorganic tin, Mance *et al* (1988d);
- Iron, Mance and Campbell (1988);
- Nickel, Mance and Yates (1984a);
- Vanadium, Mance *et al* (1988d);
- Zinc, Mance and Yates (1984b).

There are no legal limits for the protection of shellfish consumers for these contaminants in force in the UK. Therefore, except for copper, zinc and tin, these substances are not considered further here. However, maximum levels of copper, zinc and tin in food have been recommended by the Food Standards Committee (cited in FAO, 1983) and these are further outlined in Sections 6.6.1 to 6.6.3.

Zabel *et al* (1988b) reviewed the environmental effects of organotins. They concluded that the EQSs for tributyl- and triphenyltin (2 and 8 ng/l total, maximum allowable concentration) for the protection of salt water life would be adequate for the protection of human consumers of shellfish. This was on the basis that shellfish would be affected at lower levels than were likely to be harmful to humans.

6.6.1 Copper

Copper has been considered in a WRc Technical Report (Mance *et al* 1984c) and the intake from shellfish consumption was considered not to present a significant risk to human health. While there are no legal limits in force in the UK for the protection of shellfish consumers, the Food Standards Committee have recommended a maximum level of 20 mg/kg copper in food (FAO 1983). Standards for copper in shellfish for other countries are in the range 10 - 30 mg/kg, although some states in Australia may allow as high as 70 mg/kg in molluscs, (SPCC 1990).

Mance *et al* (1984c) proposed an EQS of 5 µg/l as an dissolved annual average for the protection of salt water fish and shellfish, and this has since been adopted by the DoE (1989) for the protection of salt water life. In their review, Mance *et al* (1984c) reported BCF for the mussel *Mytilus edulis* to be in the range 90 - 280, but, the highest BCF, 28 200, was reported for the American oyster *Crassostrea virginica*. It was not stated if these BCFs were for dry or wet weight. Since this review a number of BCFs for shellfish have been reported. Wet weight BCFs of 1060 and 2460 have been reported for *Crassostrea* spp., while a wet weight BCF of 7700 was reported for *Mytilus edulis*, (Wright and Zamuda 1987, Amiard-Triquet *et al*, 1988 and Calabrese *et al* 1984). The highest BCF cited in the recent literature is a wet weight BCF of 194000 for *Ostrea edulis* (Martincic *et al* 1986).

Estimating the bioaccumulation of copper in the oyster *Ostrea edulis*, in waters reaching the EQS of 5 µg/l, gives an accumulation of 970 mg/kg nearly 50 times the recommended food standard. At such a high BCF, in order to prevent the food standard being exceeded, the water concentration would need to be as low as 0.25 µg/l.

Thus while this reported BCF for *O. edulis* is nearly 100 times higher than other reported BCFs, it must be considered since oysters are commercially harvested shellfish, and a warning level of 0.25 µg/l dissolved copper is proposed. It is suggested that shellfish harvested from areas where this dissolved copper concentration is exceeded they should be checked to ensure that accumulated levels in tissues are not in excess of the recommended food standard.

6.6.2 Tin

Tin has been considered in a recent WRc Technical Report (Mance *et al* 1988b) and the intake from shellfish consumption was considered not to present a significant risk to human health. While there are no legal limits in force in the UK for the protection of fish consumers, the Food Standards Committee have recommended a maximum level of 250 mg/kg tin in food (FAO 1983), standards for tin in shellfish for other countries are in the range 50 - 250 mg/kg (FAO 1983, SPCC 1990).

Mance *et al* (1988b) have proposed an EQS of 10 µg Sn/l as a dissolved annual average concentration for the protection of salt water fish and shellfish, although this has not yet been adopted by the DoE.

However, due to lack of bioaccumulation data, it is difficult to estimate if the proposed EQS will adequately prevent the accumulation of tin in shellfish from exceeding the food guideline of 250 mg/kg.

Mance *et al* (1988b) did not report bioaccumulation factors for shellfish, and no data on BCFs since then appear to be available. They reported concentrations of tin in mollusc flesh to be in the range <0.03 - 1.2 µg/g (dry weight) but the concentration of tin in the surrounding waters was not reported. Thus while it is not possible to conclude that the EQS of 10 µg/l dissolved tin will prevent concentrations in shellfish exceeding the recommended standard, reported concentrations of tin in mollusc flesh indicate that there does not appear to be an accumulation problem, and as previously mentioned, the estimated intake of tin from shellfish is not considered a hazard, it can be suggested that the EQS will be adequate.

6.6.3 Zinc

Mance and Yates (1984b) considered that the intake of zinc from shellfish was unlikely to pose a risk to human health. While there is no legal limit in force in the UK for the protection of fish consumers the Food Standards Committee have recommended a maximum level in food of 50 mg/kg.

Mance and Yates (1984b) proposed an EQS for the protection of salt water fish and shellfish of 40 µg/l as an annual average dissolved concentration and this has since been adopted by the DoE (1989) for the protection of salt water life.

Bioaccumulation factors reported in Mance and Yates (1984b) for molluscs were in the range 43 - 16 700. The highest BCF reported was 16 700 for the oyster *Crassostrea virginica*. Bioconcentration factors for *Mytilus edulis* were in the range 282 - 500. Data reported since has shown even higher BCFs for molluscs. Martincic *et al* (1984) reported a

soft tissue concentration of 21 µg/g (wet weight) for the mussel *Mytilus galloprovincialis* after exposure to 0.0022 mg/l, this implies a BCF of 9500. Mason (1988) reported that for the periwinkle *Littorina littorea*, when exposed to 0.001 mg/l zinc, after 28 days a steady state bioaccumulation occurred, resulting in a whole body concentration of 619 µg/g zinc (dry weight) this corresponds to a BCF of 619 000 or a wet weight BCF of 30 950 (assuming ash weight is 5% of fresh weight).

Considering the highest BCF reported, 30 950 for *L. littorea*, if situated in waters reaching the EQS standard for the protection of salt water organisms, 40 µg/l zinc, the tissue concentration of zinc would exceed the statutory limit. Thus it is proposed that 1.5 µg/l dissolved zinc (which at this high BCF would give a tissue concentration of approximately 50 mg/kg) be used as a 'Warning Level' in waters for the harvesting of shellfish. This should prevent zinc concentrations in shellfish surpassing the recommended limit, in addition, it is suggested that shellfish harvested from waters where this concentration is exceeded should be checked to ensure compliance with the food standard.

6.7 Organochlorines

As previously mentioned in Section 5.7, the only organochlorine that appears to present a problem in waters used for harvesting of shellfish are polychlorinated biphenyls (PCBs). Smith *et al* (1991) reported bioaccumulation potentials of shellfish as high as 10^4 and 10^6 . In the absence of any data on safe concentrations of PCBs in food, it is suggested that the standard recommended by the EPA cited in Smith *et al* (1991) of 0.79 ng/l, be proposed as a warning level of PCBs in waters used for shellfish harvesting.

6.8 Radionuclides

Smith *et al* (1991) reported that shellfish were apparently able to accumulate radioactive material without incurring damage. As with marine fish, Ackefors *et al* (1990) reported that problems of accumulation of radionuclides would probably only occur near to discharges containing radionuclides.

The CEC (1991) (in the Directive concerned with the placing of live bivalves on the market) state that the upper limit for radionuclide content of bivalves shall not exceed limits for foodstuffs laid down by the Community. However, as for marine fish (Section 5.8), warning levels for radionuclides in water for the protection of human consumers of shellfish are not proposed due to the range of radionuclides that may be present and the limited data on the bioaccumulation potential of various shellfish. Instead it is proposed that warning levels should be based on local observations where there are known discharges of radionuclides or high background levels. Derivation of these levels should include consideration of the radionuclides present, the shellfish harvested from that area, concentrations accumulated in harvested shellfish and standards proposed by the International Commission on Radiological Protection (ICRP) for the maximum acceptable exposure via a particular pathway. The ICRP has recommended an annual dose limit of 1 mSv (milli sievert) for humans. This dose is the sum of the dose received from external radiation in one year and that received from intakes of radionuclides in the same year (NRPB 1986).

6.9 Tainting substances

As with marine fish (Section 5.9) materials should not be present in concentrations that individually or in combination produce undesirable flavours or odours in shellfish. For shellfish, ETCs have been reported for fewer substances than for fish.

The only standard that applies to shellfish in the UK has been set in the EC Shellfish Directive (CEC 1979). This sets mandatory standards for 'substances affecting the taste of the shellfish' and states that they should be at a lower concentration than that liable to impair the taste of the shellfish. Compliance with this standard is to be tested by taste. No standards from other countries are available.

As stated in Smith *et al* (1991) data on the concentrations of substances tainting shellfish are limited, with further investigations needing to be carried out. Since their review further data are not available. Therefore it is currently not possible to propose warning levels for substances in water in order to prevent the tainting of shellfish flesh. Where there is a discharge known to contain substances which may taint shellfish flesh the test procedures defined by ECETOC (1987) may provide a useful basis for setting discharge consents and monitoring compliance.

6.10 Bacteria

Bacterial contamination is a particular problem with shellfish as they are often eaten raw or only lightly cooked. Filter feeders such as oysters and mussels extract suspended solids (which includes micro-organisms) from the surrounding waters, these can then be concentrated in the mouth and alimentary tract.

The EC Shellfish Directive (CEC 1979) is concerned with the quality of shellfish waters, and applies to coastal and brackish waters designated by the Member States as needing protection or improvement in order to support shellfish (bivalve and gastropod mollusc) life and growth. This has set a guideline value of <300 faecal coliforms/100 ml in shellfish flesh and intravalvular fluid. As a note, it states that pending the adoption of a directive for the protection of consumers of shellfish products, the value should be observed in waters from which live shellfish are directly edible by man. In 1987, WHO/UNEP (Helmer *et al* 1991) recommended that shellfish areas would be considered acceptable if the faecal coliform (fc) concentration in 100 ml of shellfish flesh plus intravalvular fluid did not exceed 300.

In addition, Helmer *et al* (1991) reported that WHO/UNEP (1978) had recommended interim bacterial criteria for Mediterranean shellfish culture areas and shellfish. These included standards for the growing waters of shellfish of;

- less than 10 fc/100 ml in 80% of the samples;
- less than 100 fc/100 ml in the remaining 20% of the samples.

Bivalves have been reported to concentrate micro-organisms in their bodies up to six times the concentration in the surrounding water column (Pain 1986), but the relationship between water quality and bacterial contamination of shellfish is not constant (Evison 1985), different shellfish accumulating bacteria at different rates. Therefore the setting of a rigid bacterial standard in shellfish waters that would adequately protect human consumers is not possible.

Standards for bacterial concentrations in shellfish harvested for human consumption have also been proposed, these are outlined below.

The EC have agreed a Directive, CEC (1991), to be implemented by 1 January 1993 concerned with the placing of live bivalves on the market. The Directive states that the location and boundaries of shellfish harvesting areas (with the exception of natural scallop fisheries) must be designated by the relevant authorities (for the UK this will be MAFF) so as to identify areas from which live bivalve molluscs can be:

1. Collected for direct human consumption, such bivalves have to comply with the following bacterial requirements: less than 300 faecal coliforms or less than 230 *E. coli* per 100 g of mollusc flesh and intravalvular liquid based on a 5-tube, 3-dilution most probable number (MPN) test or equivalently precise bacteriological test; absence of salmonella in 25 g mollusc flesh.
2. Collected but only placed on the market after treatment in a purification system, after re-laying or after processing by a heat treatment sufficient to kill micro-organisms. Such live bivalve molluscs must not exceed 6000 faecal coliforms per 100 g of flesh or 4600 *E. coli* per 100 g of flesh in 90% of samples. After purification the requirements in (1) must be met.
3. Collected but only placed on the market after re-laying over a long period (at least two months) or re-laid in combination with purification systems to meet the bacterial requirements of (1) or (2), or they may be heat treated if they then meet the bacterial requirements of (1). Such live bivalve molluscs must not exceed 60 000 faecal coliforms per 100 g of flesh.

At concentrations above 60 000 faecal coliforms/100 g flesh harvesting is prohibited.

Other standards, concerning the concentration of bacteria in shellfish destined for human consumption including a more strict standard of 5000 fc/100 g for harvested shellfish which are to undergo depuration or sterilisation before consumption, are set by MAFF and DoE (Evison 1985). In addition, Helmer *et al* (1991) reported that WHO/UNEP (1978) recommend interim bacterial criteria for Mediterranean shellfish flesh of;

- 0 to 2 fc/g : sale permitted;
- 3 to 10 fc/g : temporary prohibition of sale;
- above 10 fc/g : sale prohibited.

(Shellfish in the 2 - 3 fc/g range were not mentioned)

As can be seen, the majority of bacterial standards concerning the safety of human consumers of shellfish have been for bacterial levels in the flesh, rather than in the surrounding waters. As previously mentioned, water quality criteria for bacteria in shellfish waters, in view of the fact that bacterial accumulation in various shellfish occurs at different rates, may not be suitable criteria for the protection of the health of shellfish consumers. It is therefore proposed that the criteria recommended in the Directive for producing and placing

bivalve on the market (CEC 1991), should adequately protect shellfish consumers. Thus these should be applied for shellfish intended for human consumption where possible before 1993 when they will become obligatory.

6.11 Viruses

There are currently no standards for viruses for the protection of human consumers of shellfish adopted in the UK. The EC Directive (CEC 1991) for health conditions for production and placing shellfish on the market do not include viral standards due to 'the lack of routine viral testing procedure'.

Molluscan shellfish are often eaten raw or inadequately cooked and epidemiological data has indicated that viral infections as a result of eating shellfish may occur despite bacterial standards being achieved. The absence of bacterial indicators does not necessarily mean the absence of viruses and depuration, while adequate to reduce bacterial levels sufficiently to adhere to standards, is often insufficient to rid shellfish of viruses. Gill *et al*, 1983 (cited in Cubitt 1991), reported 181 people fell ill after eating oysters that had been depurated for 72 hours. Bacterial counts were satisfactory, however depuration was inadequate in cleansing the shellfish of viruses, one oyster was enough to cause illness.

Grabow *et al* (1989), based on numerous analyses of polluted shellfish and reviewed data, concluded that current water quality criteria based on coliform indicators do not adequately detect viruses. They proposed quality standards for shellfish intended for human consumption which include standards for viruses and coliphages, Table 6.3.

Table 6.3 Recommended quality standards for shellfish intended for human consumption (Grabow *et al* 1989)

Faecal coliforms per 100 g	: 500 (90% of samples)
Faecal streptococci per 100 g	: 200 (90% of samples)
Coliphages per 100 g	: 10 (90% of samples)
Human viruses per 100 g	: 0 (90% of samples)

Total coliforms were not included in the criteria because although viruses were rarely detected in their absence, the authors found their presence was poorly correlated with that of viruses and they were present in large numbers in all samples.

Faecal coliforms, coliphages and faecal streptococci correlated well with viruses, the best correlation being with faecal streptococci. While a detection of 0 viruses per 100 g (90% of samples) was included in the standards, it was based on a 21 day cell culture technique, which would be far too long to determine if shellfish destined for human consumption adhered to the standard.

This delay in the detection of viruses in cell culture techniques is one of the main reasons for quality standards for shellfish not containing viral standards. Tissue culture may take anything from 3 to 21 days, or longer to detect enteric viruses. While other methods are

quicker they require the production of specific antibody against the virus to be detected (Gerba *et al* 1989).

However, gene probes, small strands of nucleic acid labelled with radioactive or non-radioactive markers for detection, offer a rapid and sensitive method for viral detection in shellfish. Gene probes have been developed for most of the major enteric viruses as well as viruses unable to grow in cell cultures. While they are unable to detect the infectivity of the virus, the assay time has been reported in the range 2 - 48 hours. In addition, one probe may be used to detect related viruses and since they do not rely on viral cultivation they can be used on viruses that cannot grow or be detected in cell culture (Gerba *et al* 1989).

It appears that the only way to ensure the protection of human consumers of shellfish from viral infection is to set a similar standard to that proposed by Grabow *et al* (1989) of 0 enteric viruses per 100 g. However, current techniques appear to be insufficiently quick to determine if shellfish destined for human consumption comply with this standard. Grabow *et al* (1989) found good correlation between faecal streptococci and viruses, and this should be looked into further as a possible bacterial indicator for viruses. However, with the development of gene probe techniques, adequate methods for the quick determination of viruses should soon be available. Therefore it may be possible in the future to develop practicable viral standards, but at present a viral standard for the protection of human consumers cannot be proposed.

6.12 Algal toxins

Algal toxins have recently been discussed in Smith *et al* (1991) with respect to shellfish harvesting, however, data were limited as are additional data since then. Various toxins have been identified as the cause of shellfish food poisoning, but there has been recent concern over the toxins causing paralytic shellfish poisoning and diarrhoeic shellfish poisoning. Both types of toxins, which are produced by dinoflagellate algae, accumulate in the tissues of crabs and filter feeders, these may then be eaten by human consumers. In May or June when blooms of these algae (known as red tides) may occur, particular problems may result, although detoxification when the bloom is over usually occurs within a few weeks (Ackfors *et al* 1990).

Saxitoxins (paralytic shellfish poisons, PSP), have been identified as the cause of paralytic shellfish poisoning and are produced by several genera of dinoflagellate algae. The toxins block the transmission of impulses to peripheral nerves and muscles and may cause paralysis. Recently the dinoflagellate *Gymnodinium catenatum* was identified as the organism responsible for PSP episodes resulting from consumption of mussels from a fishery in north west Spain (Anderson 1989). The dinoflagellate genus *Dinophysis* has been identified as producing poisons that cause diarrhoeic shellfish poisoning, the symptoms, diarrhoea and nausea, usually disappear within a few days.

Both the US and Canada have PSP monitoring programs (Schwalm 1989). These are based on the analysis of shellfish at key coastal stations in order to provide an early warning of increases in levels of toxic dinoflagellates in the water. However, the monitoring was found to be only partially effective due to rises in toxin levels in previously unaffected areas or during seasons when toxin levels are usually low.

The only standards that have been set in relation to algal toxins in shellfish are in the Directive (CEC 1991), due to come into effect in January 1993, concerned with placing live bivalves on the market. This sets a standard of 80 µg PSP/100 g mollusc flesh, and a non-positive result in the edible parts of molluscs for diarrhoeic shellfish poison, using 'customary biological testing methods'.

It is not possible to propose 'Warning Levels' for algal toxins in shellfish waters, since the toxins are accumulated following ingestion by filter feeders. Instead as stated by Smith *et al* (1991) if nutrient concentration are not allowed to exceed a level where algal blooms occur then there should be no associated problems of algal toxins. Cartwright and Painter (1991) suggested guideline values of 30 µg/l total phosphorus and 200 µg/l total nitrogen (annual averages) in order to prevent algal blooms in estuarine and coastal waters.

7. CONCLUSIONS

The parameters discussed in this report are those identified in previous reports as being relevant for the protection of the designated uses (Agg 1989, Smith *et al* 1991, Smith and Gulson 1991). However, information on a number of the parameters has been found to be limited especially for the protection of migratory fisheries and human consumers of marine fish and shellfish. The standards relevant to the protection of each use are presented in tables in each section and the main conclusions regarding EQSs necessary to protect each use are given below.

7.1 Salmonid fisheries

In some cases the standards set in the Fresh water Fish Directive (CEC 1978) appear to be unnecessarily stringent for the protection of salmonids. Therefore, for these parameters EQSs based on standards or guidelines recommended by other organisations have been proposed (dissolved oxygen, nitrite, BOD, chlorine). Additionally, based on the data reviewed here, two tentative standards have been proposed: a minimum dissolved oxygen concentration of 6 mg/l for the protection of spawning grounds during autumn to spring; and a maximum temperature of 20°C for waters affected by thermal discharges.

For List II substances EQSs have been proposed by WRc to DoE. Some of these standards have been adopted as national EQSs (in DoE Circular 7/89, HMSO 1989), whilst others are at various stages of discussion and consultation. The data reviewed here indicate that often there is a paucity of information for indigenous fish species. However, standards derived for the protection of 'aquatic life' or 'fish' should in general be adequate for the protection of salmonid fish.

7.2 Cyprinid fisheries

In general the same conclusions can be made for cyprinid fisheries as for salmonid fisheries. As with salmonids fisheries, in some cases the Fresh water Fish Directive (CEC 1978) appears to be unnecessarily stringent for the protection of cyprinids, and again for these parameters, EQSs based on standards or guidelines previously recommended by other organisations (dissolved oxygen, nitrite, BOD, chlorine) have been proposed.

For List II substances EQSs have been proposed by WRc to DoE, some have been adopted as national EQSs (HMSO 1989), whilst others are undergoing discussions and consultations. The reviewed data have revealed, as for salmonid fisheries, the paucity of information for cyprinid fish, especially indigenous species.

7.3 Migratory fisheries

Few standards have been derived specifically for the protection of migratory fish. Although the national EQSs for List II substances may be unnecessarily stringent for the protection of this use, compliance with the standards for less sensitive fresh water life and salt water life is mandatory in all waters receiving discharges of List II substances.

EQSs have been proposed for a number of parameters (dissolved oxygen, nitrite, BOD, chlorine, temperature and suspended solids). However, there were difficulties in proposing

standards, for migration over long distances, for DO, nitrite and temperature, due to the interaction of various other factors.

7.4 Commercial marine fish harvesting

For the protection of human consumers of commercially harvested fish, where necessary, 'Warning Levels' have been proposed. These warning levels were developed after consideration as to whether 'concentration in flesh' or 'acceptable intake' standards were likely to be exceeded in waters achieving the relevant EQSs. They are concentrations of contaminants in water which, based on the available data, are the maximum concentration that can be present in waters from which marine fish are harvested for human consumption without such standards being exceeded. The data indicate that for List II substances for which such standards exist, the EQS values for the protection of salt water life should be adequately stringent to protect this use. However, this has found not to be the case for mercury and cadmium and warning levels of 0.06 µg/l dissolved mercury and 1.4 µg/l dissolved cadmium have been proposed.

7.5 Commercial shellfish harvesting

An analysis of available bioaccumulation, food standards, and acceptable intake data to develop 'Warning Levels' indicates that EQSs derived to protect salt water life may not be sufficiently stringent to protect this use. Therefore, in order to achieve the high quality of shellfish products required, warning levels for mercury, cadmium, arsenic, lead, copper, zinc and PCB have been proposed, these are given in Table 6.2. However, since in the majority of cases data are limited, it is suggested that monitoring of concentrations in shellfish flesh should be carried out where these warning levels are implemented in order to determine if they are sufficiently stringent.

A limited number of microbiological guidelines (for faecal coliforms, enteric viruses, algal toxins) are suggested as indicators of water quality for shellfish. These have been derived from the EC Shellfish Hygiene Directive (CEC 1991). It is not possible to relate these 'in flesh' guidelines to concentrations in water because of the wide variability in accumulation potentials.

REFERENCES

Ackefors, H., Hilge, V. and Linden, O. (1990) *Contaminants in fish and shellfish products, Aquaculture Europe 1989 Business Joins Science*, edited by N. De Pauw and R. Billard, European Aquaculture Society Special Publications No. 12, Belgium.

Agg, A.R. (1989) Environmental quality standards to protect uses of fresh waters. Draft for discussion. WRc Report PRS 2189-M, WRc.

Alabaster, J.S. (1988) The dissolved oxygen requirements of upstream migrant chinook salmon, *Oncorhynchus tshawytscha*, in the lower Willamette River, Oregon, *Journal of Fish Biology*, **32**, 635-636.

Alabaster, J.S. (1990) The temperature requirements of adult Atlantic salmon, *Salmo salar* L., during their upstream migration in the river Dee, *Journal of Fish Biology*, **37**, 659-661.

Alabaster, J.S. and Gough, P.J. (1986) The dissolved oxygen requirements of Atlantic salmon, *Salmo salar* L., in the Thames estuary, *Journal of Fish Biology*, **29**, 613-621.

Alabaster, J.S., Gough, P.J. and Brooker, W.J. (1991) The environmental requirements of Atlantic salmon, *Salmo salar* L., during their passage through the Thames Estuary, 1982 - 1989, *Journal of Fish Biology*, **38**, 741-762.

Alabaster, J.S. and Lloyd, R. (1982) *Water quality criteria for fresh water fish*. 2nd Edition, Butterworths, London.

Alabaster, J.S., Shurben, D.G. and Mallett, M.J. (1979) The survival of smolts of salmon *Salmo salar* L. at low concentrations of dissolved oxygen, *Journal of Fish Biology*, **15**, 1-8.

Amiard-Triquet, C., Amiard, J.C., Berthet, B. and Metayer, C. (1988) Field and experimental study of the bioaccumulation of some trace metals in a coastal food chain: seston, oyster (*Crassostrea gogas*), drill (*Ocenebra erinacea*), *Water Science and Technology*, **20**, 13-21.

Anadu, D.I., Chapman, G.A., Curtis, L.R. and Tubb, R.A. (1989) Effect of zinc exposure and subsequent acute tolerance to heavy metals in rainbow trout, *Bulletin of Environmental Contamination and Toxicology*, **43**, 329-336.

Anderson, D.M., Sullivan, J.J., Reguera, B. (1989) Paralytic shellfish poisoning in north west Spain: the toxicity of the dinoflagellate *Gymnodinium catenatum*, *Toxicon*, **27**, No. 5, 665-674.

Anestis, I. and Neufeld, R.J. (1986) Avoidance-preference reactions of rainbow trout (*Salmo gairdneri*) after prolonged exposure to chromium (VI), *Water Research*, **20**, No. 10, 1233-1241.

Anglian Water Authority (1986) *River quality objectives*, AWA, Huntington, 1986.

Aprahamian, M.W., Strange, C.D. and Dimond, C. (1988) Movement of Atlantic salmon, *Salmo salar* L., into the River Usk, South Wales, in relation to water quality, *Journal of Fish Biology*, **33** (Supplement A), 245-247.

Baudo, R., De Bernadi, R., Diussani, G., Grimaldi, E. and Tonnolli, L. (1978) Quality objectives for aquatic life in European fresh waters, *Memorie dell'istituto Italiano di Idrobiologia*, **36**, Supplement.

Billard, R. and Robaud, P. (1985) The effects of metals and cyanide on fertilization in rainbow trout (*Salmo gairdneri*), *Water Research*, **19**, 209-214.

Birge, W.J. (1978) Aquatic toxicology of trace elements of coal and fly ash. In *Energy and Environmental Stress in Aquatic Systems*, edited by Thorpe, J.H. and Gibbons, J., National Technical Information Service, Springfield, 219-240.

Bradley, R.W. and Sprague, J.B. (1985) The influence of pH, water hardness and alkalinity on the acute lethality of zinc to rainbow trout (*Salmo gairdneri*), *Canadian Journal of Fisheries and Aquatic Sciences*, **42**, 731-736.

Brown, V.M., Gardiner, J. and Yates, J. (1984) Proposed environmental quality standards for List II substances in water, Inorganic lead, WRc Technical Report 208, WRc.

Bruton, M.N. (1985) The effects of suspensoids on fish, *Hydrobiologia*, **125**, 221-241.

Butterwick, L., De Oude, N. and Raymond, K. (1989) Safety assessment of Boron in Aquatic and Terrestrial Environments, *Ecotoxicology and Environmental Safety*, **17**, 339-371.

Calabrese, A., MacInnes, J.R., Nelson, D.A., Greig, R.A. and Yevich, P.P. (1984) Effects of long-term exposure to silver or copper on growth, bioaccumulation, histopathology in the blue mussel *Mytilus edulis*, *Marine Environmental Research II*, 253-274.

Carlson, A.R., Nelson, H and Hammermeister, D. (1986) Evaluation of site-specific criteria for copper and zinc and integration of metal addition toxicity, effluent and receiving water toxicity, and ecological survey data. EPA-600/3-86-026. Cited by the United States Environmental Protection Agency, Ambient Water Quality Criteria for Zinc, 1987.

Cartwright, N.G. and Painter, H. (1991) An assessment of the environmental quality standards for inorganic nutrients necessary to prevent eutrophication (nuisance growth of algae). WRc Report NR 2397, WRc.

CCREM (Canadian Council of Resource and Environment Ministers) (1987) Canadian Water Quality Guidelines, Inland Waters Directorate, Environment Canada, Ottawa.

CEC (Council of the European Communities) (1978) Directive on the quality of fresh waters needing protection or improvement in order to protect fish life, 78/659/EEC, *Official Journal*, **L222**, 18 July 1978.

CEC (Council of the European Communities) (1979) Directive on the quality required of shellfish waters, 79/923/EEC, *Official Journal*, L281, 10 November 1979.

CEC (Council of the European Communities) (1982) Directive on limit value and quality objectives for mercury discharges by the chlor-alkali electrolysis industry, 82/176/EEC, *Official Journal*, L81, 27 March 1982.

CEC (Council of the European Communities) (1983) Council Directive on limit values and quality objectives for cadmium discharges, 83/513/EEC, *Official Journal*, L291/1, 24 October 1983.

CEC (Council of the European Communities) (1991) Council Directive laying down the health conditions for the production and the placing on the market of live bivalve molluscs, 91/492/EEC, *Official Journal of the European Communities*, L268/1, 24 September 1991.

Chapman, G. (1986) Ambient water quality criteria for dissolved oxygen, US Environmental Protection Agency Report No. EPA 440/5-86-003, Washington DC, USA.

Chiaudani, G. and Premazzi, G. (1988) Environment and quality of life, Water quality criteria in environmental management, Commission of the European Communities, Report EUR 11638 EN.

Crawford, R.E. and Allen, G.H. (1977) Sea water inhibition of nitrite toxicity to chinook salmon, *Transaction of American Fisheries Society*, 106, No. 1, 105-109.

Crisp, D.T. (1987) Thermal 'resetting' of streams by reservoir releases with special reference to effect on salmonid fishes pp163-182 In *Regulated Streams, Advances in Ecology*, edited by Craig J.F. and Kemper J.B., 1987, Plenum Press, New York ISBN 0-306-42674-9.

Cubitt, W.D. (1991) A review of the epidemiology and diagnosis of waterborne viral infections, *Water Science and Technology*, 24, No. 2, 197-203.

Curran, J.C. and Henderson, A.R. (1988) The oxygen requirements of a polluted estuary for the establishment of a migratory salmon, *Salmo salar* L., population, *Journal of Fish Biology*, 33 (Supplement A), 63-69.

Cusimano, R.F., Brakke, D.F. and Chapman, G.A. (1986) Effects of pH on the toxicities of cadmium, copper, and zinc to steelhead trout (*Salmo gairdneri*), *Canadian Journal of Fisheries and Aquatic Science*, 43, 1497-1503.

Davis, J.C. (1975) Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review, *Journal of Fisheries Research Board of Canada*, 32, 2295-2332.

DoE (Department of the Environment/Welsh Office) (1989) Water and the environment. The implementation of European Community directives on pollution caused by certain dangerous substances discharged into the aquatic environment. Circular 7/89 (DoE), 16/89 (Welsh Office), 30 March 1989.

ECETOC (European Chemical Industry Ecology and Toxicology Centre) (1987) Evaluation of fish tainting, Technical Report No. 25, ECETOC, Brussels.

EIFAC (European Inland Fisheries Advisory Commission) (1983) Water Quality Criteria for European Fresh water Fish, Report on chromium and fresh water fish, FEW.

EIFAC (European Inland Fisheries Advisory Commission) (1984) Water Quality Criteria for European fresh water fish, Report on nitrite and fresh water fish. EIFAC technical paper 46.

Elliot, M., Griffiths, A.H. and Taylor, C.J.L. (1988) The role of fish studies in estuarine pollution assessment, *Journal of Fish Biology*, 33, (Supplement A), 51-61.

Everall, N.C., Macfarlane, N.A.A. and Sedgwick, R.W. (1989) The interactions of water hardness and pH with the acute toxicity of zinc to the brown trout, *Salmo trutta*, *Journal of Fish Biology*, 34, 27-36.

Evison, L.M. (1985) Bacterial Pollution of coastal waters in the UK and Mediterranean, *Journal of Applied Bacteriology Symposium Supplement* No. 14, 81S-93S.

FAO (Food and Agriculture Organization of the United Nations) (1983) Compilation of legal limits for hazardous substances in fish and fishery products, FAO Fisheries Circular No. 764, FAO, Rome.

FAO/WHO (1984) Codex Alimentarius Volume XVII, Contaminants, First Edition, Food and Agriculture Organization of the United Nations, World Health Organization, Rome.

Finlayson, B.J. and Vernue, K.M. (1982) Toxicities of copper, zinc and cadmium mixtures to juvenile chinook salmon, *Transactions of the American Fisheries Society*, 111, 645-650.

Friberg, L. (1988) The GESAMP evaluation of potentially harmful substances in fish and other seafood with special reference to carcinogenic substances, *Aquatic Toxicology*, 11, 379-393.

Gardiner, J. and Smith, I.N.H. (1990) Proposed environmental quality standards for sodium, chloride and sulphate in fresh water, WRc Report NR 2525, WRc.

Gerba, C.P., Margolin, A.B. and Martinez, J.H. (1989) Application of genes probes to virus detection in water, *Water Science and Technology*, 21, No. 3, 147-154.

Giattina, J.D. and Garton, R.R. (1983) A review of the preference-avoidance responses of fishes to aquatic contaminants, *Residue Reviews*, 87, 43-90.

Grabow, W.O.K., Idema, G.K., Coubrough, P. and Bateman, B.W. (1989) Selection of indicator systems for human viruses in polluted water and shellfish, *Water Science and Technology*, 21, No. 3, 111-117.

Grande, M. and Andersen, S. (1983) Lethal effects of hexavalent chromium, lead and nickel on young stages of Atlantic salmon (*Salmo salar* L.) in soft water, *Vatten*, 39, 405-416.

Gray, R.H. (1990) Fish behaviour and environmental assessment, *Environmental Toxicology and Chemistry*, 9, 53-67.

Gulson, J.D. (1992) Environmental Quality Standards to Protect Identified Uses of Controlled Waters - General Ecosystem and Special Ecosystem. WRc NRA Draft Final Report 010/3/N, WRc.

Hansan, M.R. and Macintosh, D.J. (1986) Effect of chloride concentration on the acute toxicity of nitrite to common carp, *Cyprinus carpio* L., *Aquaculture and Fisheries Management*, 17, 19-30.

Hamilton, S.J. and Buhl, K.J. (1990) Acute toxicity of boron, molybdenum and selenium to fry of chinook salmon and coho salmon, *Archives of Environmental Contamination and Toxicology*, 19, No. 3, 366-373.

Haux, C., Larsson, A., Lithner, G. and Sjöbeck, M.J. (1986) A field study of physiological effects on fish in lead-contaminated lakes, *Environmental Toxicology and Contamination*, 5, 283-288.

Hedgecote, S. (1990) Derivation of a likely safe environmental concentration for chlorine in estuarine water, WRc report.

Helmer, R., Hespanhol, I., Saliba, L.J. (1991) Public Health Criteria for the Aquatic Environment: Recent WHO Guidelines and their application, *Water Science and Technology*, 24, No. 2, 35-42.

Hermenutz, R., Hedtke, S.F., Arthur, J.W. Andrew, R.W., Allen, K.N. and Helgen, J.C. (1987) Ammonia toxicity in experimental streams, *Environmental Pollution*, 47, 249-283.

Herzig, A. and Winkler, H. (1986) The influence of temperature on the embryonic development of three cyprinid fisheries, *Abramis brama*, *Chalcalburnus chalcoides mento* and *Vimba vimba*, *Journal of Fish Biology*, 28, 171-181.

Hesthagen, T. (1989) Episodic fish kills in an acidified salmon river in south western Norway, *Fisheries*, 14, No. 3, 10-17.

HMSO (1959) Arsenic in Food Regulations, HMSO publication.

HMSO (1979) Lead in Food Regulations, HMSO publication.

HMSO (1989) The Surface Waters (Dangerous Substances) (Classification) Regulations 1989, Statutory Instrument 1989 No. 2286, HMSO London.

Huey, D.A., Simco, B.A. and Criswell, D.W. (1980) Nitrite-induced methemoglobin formation in channel catfish. *Transactions of the American Fisheries Society*, 109, 558-562.

Hugman, S.J., O'Donnell, A.R. and Mance, G. (1984) A survey of estuarine oxygen concentrations in relation to the passage of migratory salmonids, WRc Report ER 745-M, WRc.

Humpesch, U.H. (1985) Inter- and intra-specific variation in hatching success and embryonic development of five species of salmonids and *Thymallus thymallus*, *Archiv Hydrobiologie*, 104, No. 1, 129-144.

Jaffar, M. and Ashraf, M. (1988) Selected trace metal concentrations in different tissues from fish from coastal waters of Pakistan (Arabian Sea), *Indian Journal of Marine Sciences*, 17, 231-234.

Jenkins, W.R. and Willis, C.A. (1982) Environmental standards for fresh water fish/life (FT 4163C). Progress report to the Department of the Environment. 1 April to 30 September 1982, WRc Report 405-M, WRc.

Jones, A., Hedgecott, S. and Zabel, T.F. (1988) Information related to proposed 'Red List' substances, WRc Report PRU 1901-M/2, WRc.

Jones, N. and Mallet, M.J. (1985) Environmental Standards for Fresh water life III (EHT 9163 C). Report to the Department of the Environment January to June 1985, WRc Report DoE 1058-M. WRc.

Kilgour, D.M., McCauley, R.W. and Kwain, W. (1985) Modelling the lethal effect of high temperatures on fish, *Canadian Journal of Fisheries Aquatic Sciences*.

Lakshmanan, P.T. and Nambisan, P.N. (1989) Bioaccumulation and depuration of some trace metals in the mussel, *Perna viridis* (Linnaeus), *Bulletin of Environmental Contamination and Toxicology*, 43, 131-138.

Langston, W.J. (1984) Availability of arsenic to estuarine and marine organisms: a field and laboratory evaluation, *Marine Biology*, 80, 143-154.

Lewis, W.J. (1988) Uncertainty in pH and Temperature Corrections for Ammonia Toxicity, *Journal WPCF*, 60, No. 11, 1922-1929.

Lewis, Jr. W.M. and Morris, D.P. (1986) Toxicity of Nitrite to Fish: A Review, *Transactions of the American Fisheries Society*, 115, No. 2, 183-195.

Lloyd, D.S. (1987) Turbidity as a water quality standard for salmonid habitats in Alaska, *North American Journal of Fisheries Management*, 7, 34-45.

Lloyd, D.S., Koenings, J.P. and LaPerriere, J.D. (1987) Effects of turbidity in fresh waters of Alaska, *North American Journal of Fisheries Management*, 7, 18-33.

MAFF (1987) Survey of Mercury in Food. Second Supplementary Report, Food Surveillance Paper No. 17, London HMSO.

MAFF (1991) The Salmon Advisory Committee, Factors affecting natural smolt production, MAFF publications, London SE99 7TP.

Mainstone, C.P. and Wyatt, R. (1991) Development of a fisheries classification system. WRc Interim Report to NRA 244/2/Y, WRc.

- Mallet, M.J. (1986) Environmental Standards for Fresh water Life (III) Progress Report to the Department of the Environment, July 1985 to December 1985, WRc Report 1201-M, WRc.
- Mallet, M.J. (1990) Determination of Ecotoxicological Effects, Review of the toxicity of common pollutants to indigenous species of fresh water fish, I. Ammonia, arsenic and Cadmium, WRc Report NR 2541, WRc.
- Mance, G., Brown, V.M., Gardiner, J. and Yates, J. (1984b) Proposed environmental quality standards for List II Substances in Water, Chromium, WRc Technical Report TR 207, WRc.
- Mance, G., Brown, V.M. and Yates, J. (1984c) Proposed Environmental Quality Standards for List II Substances in Water, Copper, WRc Technical Report TR 210, WRc.
- Mance, G. and Campbell, J.A. (1988) Proposed Environmental Quality Standards for List II Substance in Water, Iron, WRc Technical Report TR 258, WRc.
- Mance, G., Musselwhite, C. and Brown, V.M. (1984a) Proposed environmental quality standards for List II Substances in Water, Arsenic, WRc Technical Report TR 212, WRc.
- Mance, G., Norton, R. and O'Donnell, A.R. (1988d) Proposed Environmental Standards for List A Substances in Water, Vanadium, WRc Technical Report TR 253, WRc.
- Mance, G., O'Donnell, A.R. and Campbell, J.A. (1988c) Proposed environmental quality standards for List II Substances in Water, Sulphide, WRc Technical Report TR 257, WRc.
- Mance, G., O'Donnell, A.R., Campbell, J.A. and Gunn, A.M. (1988b) Proposed Environmental Quality Standards for List II Substances in Water, Inorganic Tin, WRc Technical Report TR 254, WRc.
- Mance, G., O'Donnell, A.G. and Smith, P.R. (1988a) Proposed Environmental Quality Standards for List II Substances in Water, Boron, WRc Technical Report, TR 256, WRc.
- Mance, G. and Yates, J. (1984a) Proposed Environmental Quality Standards for List II Substances in Water, Nickel, WRc Technical Report TR 211, WRc.
- Mance, G. and Yates, J. (1984b) Proposed environmental quality standards for List II Substances in Water, Zinc, WRc Technical Report TR 209, WRc.
- Martincic, D., Numberg, H.W. and Branica, M. (1986) Bioaccumulation of heavy metals by bivalves from Limski Kanal (North Adriatic Sea). II Copper Distribution between oysters, *Ostrea edulis* and ambient water, *Marine Chemistry*, 18, 299-319.
- Martincic, D., Numberg, H.W., Stoepler, M. and Branica, M. (1984) Bioaccumulation of heavy metals by bivalves from Lim fjord (North Adriatic Sea), *Marine Biology*, 81, 177-188.
- Mason, A.Z. (1988) The kinetics of zinc accumulation by the marine prosobranch gastropod *Littorina littorea*, *Marine Environmental Research*, 24, 135-139.

McGeachy, S.M. and Dixon, D.G. (1989) The impact of temperature on the acute toxicity of arsenate and arsenite to rainbow trout (*Salmo gairdneri*), *Ecotoxicology and Environmental Safety*, **17**, 86-93.

Milne, I. and Seager, J. (1990) Proposed water quality criteria for the protection of aquatic life from intermittent pollution: dissolved oxygen, WRc Report PRS 2498-M, WRc.

Milner, N.J. (1982) The accumulation of zinc by O-group plaice, *Pleuronectes platessa* (L.), from high concentrations in sea water and food, *Journal of Fish Biology*, **21**, 325-336.

Morrison, B.R.S. (1989) The growth of juvenile Atlantic salmon, *Salmo salar* L. and brown trout, *Salmo trutta* L., in a Scottish river system subject to cooling-water discharge, *Journal of Fish Biology*, **35**, No. 4, 539-556.

Muramoto, S. (1983) Influence of complexans (NTA, EDTA) on the toxicity of nickel chloride and sulphate to fish at high concentrations, *Journal of Environmental Science and Health*, **A18**, 787-795.

Nebeker, A.V., Savonen, C. and Stevens, D.G. (1985) Sensitivity of rainbow trout early life stages to nickel chloride, *Environmental Toxicology and Chemistry*, **4**, 233-239.

Neitzel, D.A. and Becker, C.D. (1985) Tolerance of eggs, embryos and alevins of chinook salmon to temperature changes and reduced humidity in dewatered redds, *Transactions of the American Fisheries Society*, **114**, 267-273.

Nicholl, J.W., Wedemeyer, G.A., Mayer, F.L., Dickhoff, W.W., Grehory, S.V., Yasutake, W.T. and Smith, S.D. (1984) Effects of fresh water exposure to arsenic trioxide on the parr-smolt transformation of coho salmon (*Oncorhynchus kisutch*), *Environmental Toxicology and Chemistry*, **3**, 143-149.

NRPB (National Radiological Protection Board) (1986) Dose Limits for Members of the Public, Guidance on the application of protection standards, NRPB-GS4.

O'Donnell, A.R., Mance, G. and Norton, R. (1984) A review of the toxicity of aluminium in fresh water, WRc Technical Report TR 197, WRc.

Pain, S. (1986) Are British shellfish safe to eat, *New Scientist*, **111**, No. 1523, 29-33.

Persson, P.E. (1984) Uptake and Release of environmentally occurring compounds by fish, *Water Research*, **18**, 1263-1271.

Reader, J.P., Everall, N.C., Sayer and Morris, R. (1989) The effects of eight trace metals in acid soft water on survival, mineral uptake and skeletal calcium deposition in yolk-sac fry of brown trout, *Salmo trutta* L., *Journal of Fish Biology*, **35**, 187-198.

Redding, M.J., Schreck, C.B. and Everst, F.H. (1987) Physiological effects on Coho salmon and Steelhead of exposure to suspended solids, *Transactions of the American Fisheries Society*, **116**, 737-744.

- Rosemarin, A., Notini, M. and Holmgren, K. (1985) The fate of arsenic in the Baltic Sea *Fucus vesiculosus* ecosystem, *Ambio*, **14**, 342-345.
- Russo, R.C., Smith, C.E. and Thurston, R.V. (1974) Acute toxicity of nitrite to rainbow trout (*Salmo gairdneri*), *Journal of the Fisheries Research Board of Canada*, **31**, 1653-1655.
- Salveit, S J (1990) Effect of decreased temperature on growth and smoltification of juvenile Atlantic salmon and brown trout (*Salmo trutta*) in a Norwegian regulated river, *Regulated Rivers, Research and Management*, **5**, 295-303.
- Sauer, G.R. and Watabe, N. (1984) Zinc uptake and its effects on calcification in the scales of the mummichog, *Fundulus heteroclitus*, *Aquatic Toxicology*, **5**, 51-56.
- Schwalm, D.J. (1989) Shellfish sanitation challenges in the 1990s, *Water Quality Bulletin*, **14**, No. 4, 205-208.
- Seager, J., Wolff, E.W. and Cooper, V.A. (1988) Proposed environmental quality standards for List II substances in water, Ammonia, WRc Technical Report TR 260, WRc.
- Servizi, J. A. and Gordon, R.W. (1990) Acute lethal toxicity of ammonia and suspended sediment mixtures to Chinook Salmon (*Oncorhynchus tshawytscha*), *Bulletin of Environmental Contamination and Toxicology*, **44**, No. 4, 650-656.
- Siefert, R.E., Carlson, A.R. and Herman, L.J. (1974) Effects of reduced oxygen concentrations on the early life stages of mountain whitefish, smallmouth bass and white bass, *The Progressive Fish Culturist*, **36**, 186-190.
- Smith, I.N.H. and Gulson, J. (1991) Summary tables of environmental quality standards to protect identified uses of controlled waters, WRc Report NR 2920, WRc.
- Smith, I.N.H., Gulson, J. and Hedgecott, S. (1991) Environmental Quality standards to protect identified used of controlled waters, WRc Report NR 2749, WRc.
- Smith, I.N.H. and Hedgecott, S. (1990). Environmental Quality Standards to protect uses of surface waters - basic amenity (draft for discussion), WRc Report No. PRS 2517-M, WRc.
- Solbe, J.F., Cooper, V.A., Willis, C.A. and Mallet, M.J. (1985) Effect of Pollutants in fresh waters on European non-salmonid fish I: Non-metals, *Journal of Fish Biology*, **27** (Supplement A), 197-207.
- Solbe, J.F. and Shurben, D.G. (1989) Toxicity of ammonia to early life stages of rainbow trout (*Salmo Gairdneri*), *Water Research*, **23**, No. 1, 127-129.
- SPCC (State Pollution Control Commission) (1990) Water Quality Criteria for New South Wales - discussion paper, SPCC, NSW, Australia.
- Stevens, D.G. and Chapman, G.A. (1984) Toxicity of trivalent chromium to early life stages of steelhead trout, *Environmental Toxicology and Chemistry*, **3**, 125-133.

Stiff, M.J., Cartwright, N.G. and Crane, R.I. (1990) Environmental Quality Standards for Dissolved Oxygen, WRc Report PRS 2415-M, WRc.

Taylor, D., Maddock, B.G. and Mance, G. (1985) The acute toxicity of nine 'grey list' metal (arsenic, boron, chromium, copper, lead, nickel, tin, vanadium and zinc), to two marine fish species: dab (*Limanda limanda*) and grey mullet (*Chelon labrosus*), *Aquatic Toxicology*, **7**, 135-144.

Tang, Y. and Garside, E.T. (1987) Pre-exposure and subsequent resistance to lead in yearling brook trout, *Salvelinus fontinalis*, *Canadian Journal of Fisheries and Aquatic Sciences*, **44**, 1089-1091.

Thomas, R.E., Gharrett, J.A., Carls, M.G., Rice, S.D., Moles, A. and Korn, S. (1986) Effects of fluctuating temperatures on mortality, stress, and energy reserves of juvenile Coho salmon, *Transactions of the American Fisheries Society*, **115**, 52-59.

Turnpenny, A.W.H., Dempsey, C.H., Davis, M.H. and Fleming, J.M. (1988) Factors limiting fish populations in the Loch Fleet system, an acid drainage system in south west Scotland, *Journal of Fish Biology*, **32**, No. 1, 101-118.

US EPA (1986) Quality criteria for water 1986, United States Environmental Protection Agency EPA 440/5-86-001, May 1, 1986.

US EPA (1987) Ambient aquatic life quality criteria document for zinc Ref. A 3450, National Technical Information Service, United States Department of Commerce.

Van der Putte, I., Van der Galien, W. and Strik, J.J.T.W.A. (1982) Effects of hexavalent chromium in rainbow trout (*Salmo gairdneri*) after prolonged exposure at two different pH levels, *Ecotoxicology and Environmental Safety*, **6**, 246-257.

Water Quality Survey Group (1990) Proposals for formulating statutory quality objectives, A report by the Water Quality Survey Group, Draft, February 1990.

Watt, W.D. (1987) A summary of the impact of acid rain on Atlantic salmon (*Salmo salar*) in Canada, *Water, Air and Soil Pollution*, **35**, No. 1-2, 27-35.

Wedemeyer, G.A. and Yasutake, W.T. (1978) Prevention and treatment of nitrite toxicity in juvenile steelhead trout (*Salmo gairdneri*). *Journal of the Fisheries Research Board of Canada*, **35**, 822-827.

WHO (World Health Organisation) (1989) Food additives series: 24, Toxicological evaluation of certain food additives and contaminants, Prepared by the 33rd meeting of the joint FAO/WHO expert committee on food additives, pp163-219.

WHO (World Health Organisation) (1989) Lead, Environmental Aspects, Environmental Health Criteria 85, World Health Organisation, Geneva.

WHO (World Health Organisation) (1989) Mercury, Environmental Aspects, Environmental Health Criteria 86, World Health Organisation, Geneva.

WHO (World Health Organisation) (1981) Environmental health criteria 18, Arsenic. WHO, Geneva.

Wilson, W.J., Kelly, M.D. and Meyer, P.R. (1987) Instream temperature modelling and fish impact assessment for a proposed large-scale Alaska hydroelectric project, pp183-206, In *Regulated Streams, Advances in Ecology*, edited by Craig J.F. and Kemper J.B., 1987, Plenum Press, New York ISBN 0-306-42674-9.

Wilson, K., Mallet, M.J. and Whitehouse, P. (1991) Determination of ecotoxicological effects in fresh and saline waters, Review of the toxicity of ammonia, arsenic and cadmium to indigenous species of fresh water invertebrates and toxicity of ammonia to brown trout (*Salmo trutta*) Interim Report to the National River Authority, June 1990 to January 1991, WRc.

Wolff, E.W., Seager, J., Cooper, V.A. and Orr, J. (1988) Proposed Environmental Quality Standards for List II Substances in Water, pH, WRc Technical Report TR 259, WRc.

Wong, M.H., Lau, W.M., Tong, T.Y., Liu, W.K. and Luk, K.C. (1982) Toxic effects of chromic sulphate on the common carp, *Cyprinus carpio*, *Toxicology Letters*, **10**, 225-232.

Wright, D.A. and Zamuda, C.D. (1987) Copper accumulation by two bivalve molluscs: salinity effect is independent of cupric ion activity, *Marine Environmental Research*, **23**, 1-14.

Zabel, T.F., Seager, J. and Oakley, S.D. (1988a) Proposed environmental quality standards for List II substances in water, Mothproofing agents, WRc Technical Report TR 261, WRc.

Zabel, T.F., Seager, J. and Oakley, S.D. (1988b) Proposed environmental quality standards for List II substances in water, Organotins, WRc Technical Report TR 255, WRc.